River Discharge Report
SJVDIP

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This report presents the results of a study conducted by an independent Technical Committee for the Federal-State Interagency San Joaquin Valley Drainage Implementation Program. The Technical Committee was formed by the University of California Salinity/Drainage Program. The purpose of the report is to provide the Drainage Program agencies with information for consideration in updating alternatives for agricultural drainage water management. Publication of any findings or recommendations in this report should not be construed as representing the concurrence of the Program agencies. Also, mention of trade names or commercial products does not constitute agency endorsement or recommendation.

The San Joaquin Valley Drainage Implementation Program was established in 1991 as a cooperative effort of the United States Bureau of Reclamation, United States Fish and Wildlife Service, United States Geological Survey, United States Department of Agriculture-Natural Resources Conservation Service, California Water Resources Control Board, California Department of Fish and Game, California Department of Food and Agriculture, and the California Department of Water Resources.

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River Discharge Technical Committee

San Joaquin Valley Drainage Implementation Program

Final Report

January 13, 1999

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LIST OF ACRONYMS

af	Acre feet
AGR	Agricultural Supply
B	Boron
CALFED	CALFED Bay-Delta Program
CDEC	California Data Exchange Center
CDFG	California Department of Fish and Game
CDWR	California Department of Water Resources
COLD	Cold Freshwater Habitat
Cu	Copper
CVP	Central Valley Project
CVR	Central Valley Region
CVRWQCB	Central Valley Regional Water Quality Control Board
DPA	Drainage Project Area
DWR	Department of Water Resources
EC	Electrical Conductivity
EPA	Environmental Protection Agency
GBP	Grassland Bypass Project
IND	Industrial Service Supply
IRDROP	Irrigation and Drainage Operations
IRIS	Integrated Risk Information System
kg	Kilogram
LBNL	Lawrence Berkeley National Laboratory
LOEC	Lowest observable effect concentration
lbs	Pounds
MAA	Management Agency Agreement
mg	Milligram
MIGR	Migration of Aquatic Organisms
Mo	Molybdenum
MOU	Memorandum of Understanding
MP	Management Plan
MUN	Municipal and Domestic Supply
NAV	Navigation
NOAEL	No Observed Adverse Effect Level
NOEC	No observable effect concentration
ppb	Parts per Billion
ppm	Parts per Million
PRO	Industrial Process Supply
REC-1	Water Contact Recreation
REC-2	Non-Contract Water Recreation
RWQCB	Regional Water Quality Control Board
Se	Selenium
SGWD	South Grassland Water District
SJR	San Joaquin River
SJRIO	San Joaquin River Input Output

SJRIODAY	San Joaquin River Input Output Daily Model
SJRMP-WQS	San Joaquin River Management Program-Water Quality Subcommittee
SJV	San Joaquin Valley
SJVDIP	San Joaquin Valley Drainage Implementation Program
SJVDP	San Joaquin Valley Drainage Program
SLD	San Luis Drain
SPWN	Spawning, Reproduction, and/or Early Development
SWRCB	State Water Resources Central Board
TDS	Total Dissolved Solids
TMML	Total Maximum Monthly Load
UA	Use Agreement
ug/L	Microgram per Liter
USBR	United States Bureau of Reclamation
USBR-CVO	United States Bureau of Reclamation-Central Valley Operations
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WARM	Warm Freshwater Habitat
WDR	Waste Discharge Requirements
WILD	Wildlife Habitat

1. Introduction

The westside San Joaquin Valley is a fertile yet arid landscape where commercial agriculture is viable only with supplemental irrigation. Irrigation water from either surface or underground sources, has naturally accumulated salt and other minerals from contact with the earth. The efficient application of irrigation water sufficient to meet crop demands and leaching to remove salt from the crop root zone to maintain soil quality, results in the deep percolation of applied water. Infiltrating irrigation water dissolves and leaches salts and trace elements into the shallow groundwater, a necessary consequence of maintaining a salt balance in the crop root zone. When irrigation water is applied without the provision of adequate drainage, a shallow water table on low permeability clay rises toward the soil surface, water-logging the crop root zone, and leaving salts and trace elements to accumulate in the rootzone as a result of crop evapotranspiration.

The San Joaquin River has historically provided essential drainage for both agricultural land and managed wetlands in the Grassland Basin of the San Joaquin Valley. The Grassland Basin is comprised of several agricultural water and drainage districts, several federal and state managed wildlife refuges, and a large area of private duck clubs. To the south, the Tulare and Kern Basin region of the San Joaquin Valley has no drainage outlet to the San Joaquin River, except in times of extreme flood. Where drainage discharge is available, the drainage water typically contains high concentrations of dissolved solids, and some trace elements, particularly selenium and boron. Most of the selenium and boron load contained in drainage water originates from resident groundwater displaced into drain lines by infiltrating irrigation water. Groundwater concentrations of salts and trace elements are generally considerably higher than the leachate concentrations. The major water quality problems in the San Joaquin River are caused by the high loadings of salt, selenium, and boron in the displaced groundwater discharged to the River. Both federal and State water quality objectives have been developed to protect fish and wildlife, to protect riparian agricultural irrigation diverters in the South Delta, and to protect municipal and industrial water agencies and users that divert water from the Delta.

The 1990 Management Plan developed by the San Joaquin Valley Drainage Program recommended a number of drainage management measures to be implemented in the Grassland Basin, including source control (reduction in applied irrigation water), reuse of drainage on salt tolerant plants, evaporation ponds, land retirement, and groundwater management. The report also recommended a continuation of limited discharge of drainage to the San Joaquin River, while meeting water-quality objectives, specifically for selenium and boron at Crows Landing. The discharge was to be conveyed to the River in a reopened portion of the San Luis Drain with an extension to the San Joaquin River below its confluence with the Merced River, for the purpose of maximizing the benefit of the dilution capacity of the Merced River inflow.

Since 1990, local growers have made a number of advances in drainage reduction, primarily in the area of source control. Until about two years ago, drainage discharged from agricultural lands passed through a network of channels in the Grassland wetlands to Mud Slough North and Salt Slough to enter the San Joaquin River. Starting in 1996, implementation of the Grassland Bypass Project has consolidated agricultural subsurface drainage flows on a regional basis, and reopened a portion of the San Luis Drain to redirect drainage flow from the wetland areas to Mud Slough (north) and then the San Joaquin River, thereby effectively removing subsurface drainage from all but Mud Slough North. The Grassland Bypass Project specifies selenium load limits on monthly and annual basis with the specification that annual selenium loads be reduced by 5 percent each of project years 3 through 5. Major issues since implementation, such as the initial failure to achieve selenium load targets, in large part a result of unusually high precipitation and flood events, and questions concerning the biological effects of the rerouted drainage remain to be resolved.

The Discharge to the River Technical Committee was convened by the San Joaquin Valley Drainage Implementation Program to assess the status and need for updating the 1990 Management Plan River discharge recommendation for the Grassland Basin subarea. Other technical committees were established to review the other recommendations applicable to the Grassland Subarea, the scope of this report is biological impacts and management options of drainage discharged off-farm.

The report begins with a description of San Joaquin River and Grassland Basin watershed hydrology and presents an overview of data on drainage discharge and water quality from 1985 up to implementation of the Grassland Bypass Project in 1996, identifying salt and trace element constituents and sources. The latest scientific information on the reactions and ecotoxic effects of selenium, boron, and molybdenum in aquatic systems is then reviewed. The next section summarizes presently established water quality objectives and beneficial uses for lower San Joaquin River and westside tributary water. This is followed by a synopsis of the 1990 Management Plan recommendations on drainage discharge. The next section details developments since the 1990 Plan, focusing on the Grassland Bypass Project, the preliminary results of monitoring water quality and biological effects of the Project, and the status of developing a system of real-time drainage management that would match the timing of drainage discharge to the assimilative capacity of the San Joaquin River. The final section on current assessment and new recommendations focuses on the major unsolved issues of the potential for implementing real-time management and the determination of the site-specific ecotoxicity of selenium.

II Existing Hydrology and Water Quality

II.A. Watershed Hydrology

The San Joaquin River (SJR) drains a watershed of approximately

35,000 square kilometers (13,500 square miles). Snow melt and rainfall in the upper eastern watershed, the Sierra Nevada mountains and foothills, is the major source of runoff to the SJR. During the period 1985 to 1994, roughly an average of 70 percent of the annual flow in the SJR was from three major east-side tributaries, the Stanislaus, Tuolumne, and Merced Rivers, as well as the mainstem. The remainder of the flow during this period consisted of tailwater (12 percent), subsurface drainage (1 percent), and two minor westside tributaries, Salt Slough and Mud Slough North (10 percent), and groundwater accretions (4 percent) (Figure II-3).

According to USGS and USBR records, the mean annual discharge of the SJR near Vernalis was approximately 3.3 million af per year between 1930 and 1997. Variations in annual discharge during this period ranged from just over 400,000 af in Water Year 1977 to over 15 million af in Water Year 1983. Unimpaired runoff in the Basin would be considerably greater. Much of the natural flow is now stored, diverted and consumptively used before reaching Vernalis. Eastside rainfall and snow melt runoff is regulated and stored in a series of reservoirs in the eastern watershed. There are 57 major reservoirs, four of which can store over one million acre-feet of water. The four largest reservoirs that control flow on the eastside tributaries and mainstem are:

1. New Melones Reservoir on the Stanislaus River,
2. New Don Pedro Reservoir on the Tuolumne River,
3. Lake McClure on the Merced River, and
4. Millerton Lake on the mainstem SJR.

Stream flow in the Lower SJR (from Mendota Pool to Vernalis) is reduced by diversions and increased by agricultural and wetland return flows. In summer and fall months, smaller streams in the SJR system and portions of the River above Sack Dam to Mendota Pool frequently have little or no flow. The majority of the River flow upstream of the Merced River to Sack Dam during low flow periods is composed of agricultural and wetland return flows and groundwater accretions.

The primary land use in the lower watershed, the San Joaquin Valley, is irrigated agriculture. Most of the water supply for irrigated agriculture on the west-side of the Valley is imported from the Sacramento-San Joaquin Delta, whereas the east-side tributaries plus the SJR at Friant and groundwater provide the majority of the water supply to east-side of the Valley (see Figure II-1).

FIGURE II-1

II.B. Drainage Discharges and Water Quality, 1985-1995

The Grassland Basin, bounded by Westlands Water District to the south and State Highway 140 to the north, SJR to the east, is a hydrologic subunit of the SJR watershed. The soils in the Grassland Basin have low permeability and are naturally high in salts. The low soil permeability combined with the importation of water has resulted in formation of a shallow groundwater table. To maintain productivity, it has become necessary to install artificial drainage in low lying agricultural areas. Subsurface drainage, produced from a 41,000 hectare (101,000 acres) agricultural area in the southern part of the Grassland Basin (hereafter referred to as the Drainage Project Area - DPA), is high in soluble salts and also contains significant concentrations of certain trace elements that can be harmful to fish and wildlife. The primary trace elements of concern are selenium and boron.

In addition to discharges from the DPA, discharges from surrounding wetland areas during the spring months also contribute a significant salt load to the SJR. Prior to the Grassland Bypass Project, the combined discharges from the agricultural lands and wetlands were conveyed through a system of canals and natural streams to the SJR. The salt load contribution to the SJR from the combined agricultural and wetland discharges from the Grassland Basin is high relative to other sources of salt in the Basin. Downstream dilution of poor quality discharges from the Grasslands Basin is provided by east-side tributaries. Flows in the east-side tributaries are largely regulated by upstream reservoirs which, in turn, are operated according to set rules and release schedules. These rules and release schedules are based on the combined demand for flood storage, fish migration pathways, irrigation supply, hydropower, water quality maintenance, and recreation.

In contrast to the high degree of regulation and control of eastside tributary flows, the discharge from the DPA has historically been largely uncontrolled. Sump pumps associated with subsurface agricultural drainage systems were designed to turn on automatically when the water reached a set level in the sump. Hence, the pattern of discharges from agricultural lands has generally mirrored the irrigation season. In contrast, surface drainage discharge from seasonal wetlands occurs in early spring between February and April. Some control of the discharge release schedule for seasonal wetland drainage can be exercised by wetland managers, although these schedules are determined to a large extent by habitat requirements, and management preferences of State & Federal refuge managers and local, privately owned duck clubs.

Discharges from Mud Slough (north) and Salt Slough have been high in dissolved salts and trace elements, including selenium, boron and/or molybdenum, when dominated by west side agricultural subsurface drainage. High concentrations of salts in the lower reaches of the SJR have occurred downstream of Salt Slough when the flow was largely composed of both subsurface agricultural drainage and groundwater accretions. As a result, uses of Lower SJR water have been impacted due to poor water quality. Typically water quality in the east side tributaries of the Merced, Tuolumne and Stanislaus Rivers is good to excellent, and dilutes salinity from west side irrigation return flows, wetland discharges, and groundwater influx to the River.

Figure II-2 shows salt concentration as EC for the Lower San Joaquin River at the farthest downstream sampling point (above the Delta) near Vernalis. From October 1985 to September 1996, 30-day running averages ranged from approximately 150 to 1,400 _mhos/cm (additional water quality information for this time period can be found in Steensen et al.1998). The water quality objectives (see section IV) of 700 mS/cm from April through August and 1000 mS/cm from September through March are depicted by horizontal lines on Figure II-2. Note that the objectives have been frequently exceeded but there are time periods when more salt could be accommodated without exceeding the water quality objective. The water quality objective could be achieved more frequently if the timing of all the salt inflows were altered (see section VI.D. Real-Time Management of Drainage Discharge). However, groundwater accretions are essentially uncontrollable.

The mean annual water discharges to the SJR for water years 1985 to 1995 from various sources are presented in Figure II-3. East-side tributaries contribute about 75 percent of water flow in the river. The flows by years from the various sources are presented in Figure II-4 which illustrates the high annual variability. The greatest annual variability occurs with the east-side tributaries, and reflects the variability in annual precipitation. The annual variability of water discharges to the SJR are much less for other sources as compared to the east-side tributaries.

The mean annual salt loadings to the SJR for years 1985 to 1995 from various sources are illustrated in Figure II-5. Mud and Salt Sloughs contributed about 43 percent of the total salt load. The contributing sources of salt to Mud and Salt Sloughs are shown in Figure II-6 and are relatively equally divided among the sources. About 50 percent of the salt load to Mud and Salt Sloughs originated from the DPA. Therefore, only about 22 percent of the total salt load to the lower SJR originated from the DPA. The annual variations in salt discharges are shown in Figure II-7. Salt loads are highest in the years (1986 and 1995) of highest river discharge (Figure II-4).

On average, about 64 percent of the SJR selenium load has been contributed by Mud and Salt Sloughs (Figure II-8). The annual variability of selenium loads to the River is illustrated in Figure II-9. The total annual water discharge, salt loads and selenium loads (Figures II-4, II-7, and II-9) follow a similar pattern with 1986 and 1995 the years of highest discharge and loads, and1991 and 1992 the lowest years.

The comparison of water, salinity and selenium mean annual (1985-1995) discharge from six sources are depicted in Figure II-10. Mud and Salt Sloughs have contributed relatively low water volumes, moderately high salt loads, and very high selenium loads as compared to the east-side tributaries.

By way of transition to the post-Grassland Bypass Agreement water quality data presented in section VI.C., Figures II-11 through II-14 show the Water Years 1986-1997 annual discharge, TDS, boron, and selenium loads for two SJR sites, Crows Landing and Vernalis, and two additional Grassland Watershed sites, one summarizing loads from DPA and the other depicting loads from the entire watershed. In sequence from DPA, to the Grassland Watershed, to Crows Landing, and finally to Vernalis, the sampling sites are located in a downstream succession with increasing annual discharge or flow volumes as shown in Figure II-11. The loads of salt and boron also increase in succession from upstream to downstream sampling sites (Figures II-12 and II-13). Water Years 1996 and 1997 were wet years with similar discharge and loading to Water Years 1986 and 1995. Although annual discharge at Crows Landing and Vernalis were higher in 1997 than during all other years, including 1995, loads of salt, boron, and selenium were higher at both sites in Water Year 1995. Loads of selenium at both SJR sites were lower in 1997 than in both 1995 and 1996, even though discharge was highest in 1997. Highest loads of all constituents occurred in 1995 and may be partially explained from the leaching of salts that had accumulated in the Grassland Basin during preceding eight years of below average rainfall, water supplies, and discharge.

FIGURE II-2

FIGURE II-3

FIGURE II-4

FIGURE II-5

FIGURE II-6

FIGURE II-7

FIGURE II-8

FIGURE II-9

FIGURE II-10

FIGURE II-11 and FIGURE II-12

FIGURE II-13 and FIGURE II-14

In contrast, selenium loads at the drainage project are equal to or greater than at Vernalis on eight of the twelve years.

A review of currently available scientific information on the biogeochemistry and ecotoxicity of selenium, boron and molybdenum is presented in section III. Molybdenum data collected in the Grassland Basin and the SJR watershed are shown in Table III-2.

III. Constituents of Concern - Current Status

The primary constituents of concern from agricultural subsurface drainage discharges to the San Joaquin River are salinity, selenium, boron, and molybdenum (SJVDP, 1990a). These constituents may be beneficial to plant and/or animals in smaller concentrations, but can be toxic at higher concentrations. Salinity, measured as total dissolved solids or electrical conductivity, can affect a number of beneficial uses, particularly irrigated crops. The primary concern of selenium is its accumulation through the food chain and ecotoxic effects. Boron is a concern to a number of beneficial uses but is regarded as primarily a problem to agricultural crops. Molybdenum is toxic to livestock, particularly ruminants, such as cattle.

III.A. SALINITY

Water quality criteria for the San Joaquin River system should be based on uses and species found in the SJR system. Specific agricultural crops have been documented as sensitive to low salt concentrations with most sensitive crops affected at concentrations in the soil water at 1.6 dS/m (TDS of 1,040) or lower. Irrigation may dilute or may concentrate salts in the soil depending on the salinity of the irrigation water, leaching, evapotranspiration and other factors. For industrial purposes, criteria documents show a wide range of acceptable levels of TDS concentrations (from 150 to 118,000 mg/L). In a consumer survey of drinking water tastes, TDS concentrations of greater than 1,300 mg/L were considered unacceptable. Fish, specifically squawfish, chub, and bonytail, avoid water with TDS ranging from 4,400 to 6,600 mg/L. The effects of salinity on poultry and livestock drinking water vary, however negative effects are not seen below 3,000 mg/L. Livestock and poultry can adjust to gradual increases in concentrations of salts, but a sudden increase from slightly to highly mineralized waters causes acute distress of varying severity. It follows that criteria to fully protect irrigated agricultural crops is lower than the criteria to protect livestock, fish and wildlife.

Municipal and Domestic Water Supplies

In a study of water with TDS values that ranged from 100 to 2,300 mg/l, consumer acceptance decreased as mineral content increased (USEPA, 1973). USEPA (1973) recognized that a large number of water supplies that contained dissolved solids with concentrations greater than 500 mg/L were used without obvious side effects. Consumer survey, conducted in the 1960's (USEPA 1976; 1986) in

29 California water systems using taste as a criterion resulted in the following classifications based on taste thresholds of dissolved salts:

319 to 397 mg/L dissolved solids as "excellent,"
658 to 755 mg/L as "good," and
1,283 to 1,333 mg/L as "unacceptable."

The State of California secondary drinking water standard is 500 mg/L, and the USEPA taste, odor and welfare standard is 250 mg/L (Marshack, 1998).

Industrial

According to McKee and Wolf (1963), dissolved solids in industrial waters can result in foaming inside boilers and interfere with clearness, color, or taste of many finished products. Elevated salt concentrations also can accelerate corrosion. USEPA=s (1973) Water Quality Criteria document summarizes characteristics of surface waters that have been used as sources of industrial water supplies. Maximum concentrations of dissolved solids ranged from 150 mg/L used in the textile industry to 118,524 mg/L used in oil recovery injection waters. A maximum of 1,000 mg/L was used for cooling water. The chemical and petroleum industry used maximum concentrations of 2,500 and 3,500 mg/L, respectively. The primary metals industry used 1,500 mg/L of dissolved solids.

Fish and Other Aquatic Life

Published water quality information related to fish and aquatic life forms do not always clearly separate the effects of salinity from the effects of specific toxic elements. The number biological species for which there is information is limited. However, some ecotoxicological research can be summarized in general and specifically for the San Joaquin River. The SJR is a more sulfate dominated river system than ocean/bay water which is more chloride dominated. Certain biota can be expected to respond differently to the varying anionic systems.

Dwyer, *et al.* (1992) did an analyses of toxicity on striped bass *(Morone saxatus)* and water flea *(Daphnia magna)* using irrigation drain water entering Stillwater Wildlife Management Area, Nevada. *D. magna* showed 100 percent mortality (or immobility) in test waters at 10,000 to 11,500 mg/L of TDS after 48 hours of exposure. Striped bass were more tolerant of highly saline waters than *D. magna.* Their tolerance was inversely related to the hardness of the water. Lethal effects of salinity on striped bass depended on the specific ionic composition of the saline water. A mixture of trace elements was toxic to striped bass although individual elements were below expected acutely lethal concentrations. The authors concluded that salinity is an important water quality characteristic, but the ionic composition of water must also be considered.

According to USEPA (1976; 1986), all species of fish and other aquatic life must tolerate a range of salinity to survive in nature. Studies in Canada (Saskatchewan) indicate that many common freshwater species survived in water of 10,000 mg/l dissolved solids, but only two species, whitefish and pike-perch survived 15,000 mg/L. Only one species, stickleback survived 20,000 mg/L dissolved solids. Dissolved solids in excess of 15,000 mg/L were considered unsuitable for freshwater fish.

The San Joaquin Valley Drainage Program (1990a) reviewed the biological effects of total dissolved solids from agricultural drainage. In a 1983 study by Pimentel and Bulkley, preliminary experiments had shown juvenile chubs to be more sensitive than larger fish. Squawfish, chub and bonytail preferred TDS ranges of 560 - 1,150, 1,000 - 2,500 and 4,100 - 4,700 mg/L, respectively. The same species of fish avoided high TDS concentrations of 4,400, 5,100 and 6,600 mg/L, respectively. Low temperature reduces the ability of fish to regulate internal salt balance. Hence, salinity concentrations that were acceptable in the summer months were avoided in the winter.

According to Saiki, et al. (1992), survival of Chinook salmon was significantly reduced by exposure to 100 percent tile drain water with TDS that ranged from 12,000 to 18,000 mg/L. They evaluated the dilution effect of San Joaquin River water collected at Crows Landings Road and dealt with toxicity of trace elements. Chinook salmon that survived 28 days of exposure were analyzed for trace elements (selenium and boron). Concentrations of these trace elements in the whole bodies of these fish increased as the percentage of drain water increased. TDS ranged from 900 to 1,400 mg/L in the River and from 18,000 to 23,000 mg/L in drain water.

Survival of striped bass was also affected by exposure to tile drain water, but not by exposure to reconstituted sea water (Saiki, et al., 1992). They stated that fish in reconstituted sea water, which had concentrations of TDS similar to the tile drainage water but with chloride instead of sulfate as the dominant anion, survived and developed well. They concluded that the toxicity of tile drain water is not a simple effect of excessive concentrations of dissolved salts. Salmon and striped bass were not able to tolerate the unusual ratios of major cations, anions and/or sulfates. They recommended considering potential toxicity from unusual ratios of major ions and high concentrations of sulfate. In a partial confirmation of this concept, twenty-eight day exposure tests were run using waters with different mixtures and concentrations of salts (Saiki, et al., 1992) using juvenile Chinook salmon and striped bass. The fish survived and grew well in reconstituted sea water with TDS concentrations similar to those in the agricultural drainage water, but with chloride instead of sulfate as the dominant anion. However, fish in drainage water at lower salt concentrations (50 percent dilution) experienced either death or poorer growth.

Although Saiki=s research implies increased toxicity in a sulfate dominated riverine system, there is further data on the singular effect of sulfate compared with other chemicals. Mount and Gulley (1992) developed relationships for salinity that predict acute toxicity of saline waters to freshwater organisms based on major ion composition. They focused on seven major ions (sodium, potassium, calcium, magnesium, chloride, sulfate and bicarbonate) and three freshwater test species (two water fleas, *Ceriodaphnia dubia and Daphnia magna,* and a flathead minnow*, Pimephales promelas).* The sulfate ion was the least toxic and potassium was the most toxic of all the ions analyzed. Sodium and calcium did not appear to be directly toxic to any of the test species. Results were based on laboratory toxicity tests on over 3,000 combinations of major ions and the development of multivariate regression equations that related major ion concentrations to survival of the three test species.

Juvenile salmon outmigrating from the Merced, Tuolumne and Stanislaus River's are moving from a freshwater environment to the San Joaquin River (high in sulfates), to the south Delta (fresher water) to the bay (chloride dominated). These outmigrants are also undergoing physiological changes during smoltification that will allow them to adapt to marine environments. There is no information available on the effects of sulfates from the San Joaquin River on juvenile Chinook salmon during outmigration and the smoltification process. It is also unknown how long these smolts remain in the San Joaquin River. Analyses, to date, suggest that mortality during the "smolt" life stage is an important determinant in adult production in the San Joaquin Drainage (CDFG, 1991). Research is needed to determine the effects of salinity in the San Joaquin River on Chinook smolts and rearing time in the River. Adult salmon migration into the San Joaquin River is triggered by fresh outflow from the eastside tributaries. The timing of adult Chinook salmon migrating upstream to spawn may be delayed due to high salinity.

Crop Use

Summaries of the effects of salinity on crops have been published by Mass (1990) and Shalhevet (1994). The 1987 SWRCB Technical Committee report states that salinity is a concern primarily because of the effect of salts on irrigated crops, the most sensitive beneficial use.

The abstract of the paper Shalhevet (1994) is an excellent and short summary of these effects. The following are the key sentences that apply to salinity impacts on irrigation management pertinent to this report: Note I=ve reduced several words from caps to noncap in the last sentenced

> ***ADuration of exposure and stage of growth****: plants are more sensitive during the seedling stage than during later stages of growth. But the preponderant*

temporal effect of salinity is the duration of exposure. ***Spatial distribution:*** *the best estimate of the effective salinity when salt is non-uniformly distributed with depth is the mean salinity within the root zone.@ A****Soil fertility:*** *the level of soil fertility has no effect on the tolerance of crops to salinity.* ***Varietal differences:*** *differences in salt tolerance among varieties exist mainly in fruit trees, which are specifically sensitive to chloride and sodium salts. Differences among field and garden crops are not common and are usually small.@ ALeaching requirement: leaching is the key to the successful use of saline water for irrigation. Under normal field conditions with free drainage the leaching provided by the normal inefficiencies in irrigation should be sufficient to control salinity. When leaching is necessary, it should be provided at the time when the soil salinity reaches hazardous levels.* ***Irrigation frequency:*** *the bulk of the evidence shows no advantage to increasing irrigation frequency when saline water is used, except possibly under excessive leaching.@ A****Availability of more than one water source:*** *blending of saline with non-saline water is a questionable practice. It is preferable to use the non-saline water source early in the growing season and the source of saline water successively.* ***Irrigation method****: drip irrigation, where feasible, gives the greatest advantages when saline water is used. Sprinkler irrigation may cause leaf bum on sensitive crops. The damage may be reduced by night irrigation and by irrigating continually rather than intermittently.* ***Drainage:*** *the critical depth to the water table is determined mainly by the aeration requirement of the crop, as long as a net downward flux of water is maintained by natural or properly designed man made drainage system.@* ***ASoil hydraulic conductivity (K) and drainable porosity:*** *important parameters in drainage design, are strongly influenced by the composition and concentration of the irrigation water. The higher the sodium adsorption ratio (SAR), the greater the reduction in K. The detrimental effect of high SAR is mitigated as the total salt concentration increases.@*

Maas (1990) has developed a number of tables relating salt tolerance to crop type. The soil salinity criteria for the relative salt tolerance classification, based on the electrical conductivity of the water obtained from a saturated soil paste (ECe), are given in the table below for 100 percent yields.

Soil Salinity Criteria For The Classification Of Relative Salt Tolerance (Maas, 1990)

Salt Tolerance Classification	Classification Abbreviation	Range in ECe, dS/m
Sensitive	S	0 B 1.6
Moderately Sensitive	MS	1.6 B 3.2
Moderately Tolerant	MT	3.2 B 6.3
Tolerant	T	6.3 B 10.2

Since these concentrations are for soil water extracts, the addition of less saline water through irrigation or rainfall can reduce the effects of salinity in San Joaquin Valley surface waters. Irrigation water may dilute or may concentrate salts in the soil depending on applied water quality, leaching, evapotranspiration, porosity and other factors.

The relative salt sensitivity for crops in the Lower San Joaquin Valley vary from sensitive crops, such as bean and tree crops (eg. apricots), to moderately sensitive crops, such as alfalfa and broccoli. The most salt tolerant crop grown in the Lower San Joaquin Valley is cotton. Sweet corn is moderately sensitive. The SJVDIP Drain Water Reuse Technical Committee report provides more detailed information about the effects of salinity on crop productivity, soil quality, and crop production systems.

Poultry and Livestock Drinking Water

Much of the research on the effect of salinity in poultry and livestock drinking water was performed prior to 1960, and has been summarized in McKee and Wolfe (1963). Multiple and sometimes conflicting effects for a given level of salinity for the same animal species are evident. McKee and Wolf (1963) reported the effects of various concentrations of total salts on livestock that included Ainjurious@ effects as low as 3,000 mg/L, but cited a reference that showed cattle thrived at 18,000 mg/L. Poultry and livestock can be injured by drinking water that contains excessive dissolved solids. Weakness, reduced milk or egg production, bone degeneration, and death may result from highly saline waters.

Animals can temporarily drink highly saline water without harmful effects. Animals can also adjust to gradual increases in concentrations of salts, but a sudden increase from slightly to highly mineralized waters causes acute distress and diarrhea of varying severity. The ability to adapt to saline water depends on the kind of salts present, species of animal, diet, age, physiological condition, season of year, climate, and other factors.

More recent USEPA criteria publications (1976; 1986) state that chickens, swine, cattle, and sheep can survive saline waters with up to 15,000 mg/L of sodium and calcium combined with bicarbonates, chlorides and sulfates. However, these animals could survive on only I0,000 mg/L water with only salts of potassium and magnesium. The 1987 SWRCB Technical Committee report states that livestock are able to tolerate somewhat more saline water without severe effects than irrigated crops.

The USEPA (1973) *Water Quality Criteria* document summarizes several studies on the effects of saline waters on beef heifers in Nevada. Waters containing about 5,000 mg/L or less of sodium sulfate showed no effects of specific ions, but heifers drank less, lost weight and had increased methemoglobin and sulfhemoglobin levels.

Wildlife

For wildlife, toxicity depends on many factors, such as concentration of the compound, ability of the organism to acclimate, individual ions present, availability of alternative freshwater habitats, and the age of the organisms (SJVDP, 1990a). Different ions (sodium, potassium, calcium and magnesium) in highly saline waters cause different toxic effects on animals. Factors affecting the salt gland in birds can affect survival of ducks in highly saline water (SJVDP 1990a). Ducklings begin secreting saline fluids from the salt gland about 6 days after birth and require fresh water until that age. Marine birds generally have far larger and more developed salt glands than inland birds. Research suggests that ducks suddenly restricted to high salt concentrations in water, such as during a drought following wet years, are at the greatest risk of salt toxicity than those continuously exposed to salty water.

The SJVDP final report (I990b) states that freshwater ducklings were very sensitive to salty water. Toxicity tests showed that molt was delayed in mallard ducklings that drank water with a TDS concentration of 3,000 mg/l, and growth was reduced when drinking water with an EC of 7.7 dS/m, which has an estimated TDS of 5,400 mg/l.

Swanson *et al.* (1984) investigated duckling use of North Dakota saline lakes dominated by sodium, magnesium and potassium sulfates. In field studies, five species of ducklings that were less than 3 days old, experienced some mortality when exposed to waters with a specific conductance of 16.0 dS/m (TDS of 10,400 mg/L). The ducklings could not tolerate salt concentrations that exceeded 20.0 dS/m (TDS of 13,000) unless a supply of fresh water was also available. Fifty percent mortality occurred after one day of exposure.

III.B. Selenium

1. Importance of Selenium Biogeochemical Transformations Through the Foodweb.

Through decades of research efforts since the incidents of Belews Lake, NC (recipient of coal-fly ash) and Kesterson Reservoir (recipient of agricultural drainage waters), it is clear that Se biotransformations into organo-selenium forms and their accumulation by the aquatic foodweb hold the key to the understanding of Se ecotoxicology (Adams et al., 1997a; Saiki et al., 1993). Since mitigating Se impact on wildlife is one of the ultimate goals of the San Joaquin Valley Drainage Implementation Program (SJVDIP), it is critically important to address Se biotransformations and biotransfer through the aquatic foodweb and to utilize the information to guide management decision. In light of the emerging evidence that some intrinsic or natural attenuation of waterborne Se is occurring in agricultural evaporation basins of the San Joaquin Valley (Fan and Higashi, 1998; Fan et al., 1998), a mechanistic understanding of how Se is biotransformed by biota, particularly by primary producers, may directly help alleviate the drainage problems. However, details of Se biotransformation in aquatic environments, particularly in the San Joaquin River, are severely lacking.

2. Current Understanding of the Selenium Biogeochemical Cycle in Aquatic Systems

Our present understanding of Se biogeochemistry can be represented in the following scheme (Figure III-1). It is important to keep in mind that this scheme is largely derived from analogy to the sulfur biogeochemical cycle and that much of the details of this cycle, particularly regarding the biotransformation pathways in the biota, has yet to be elucidated.

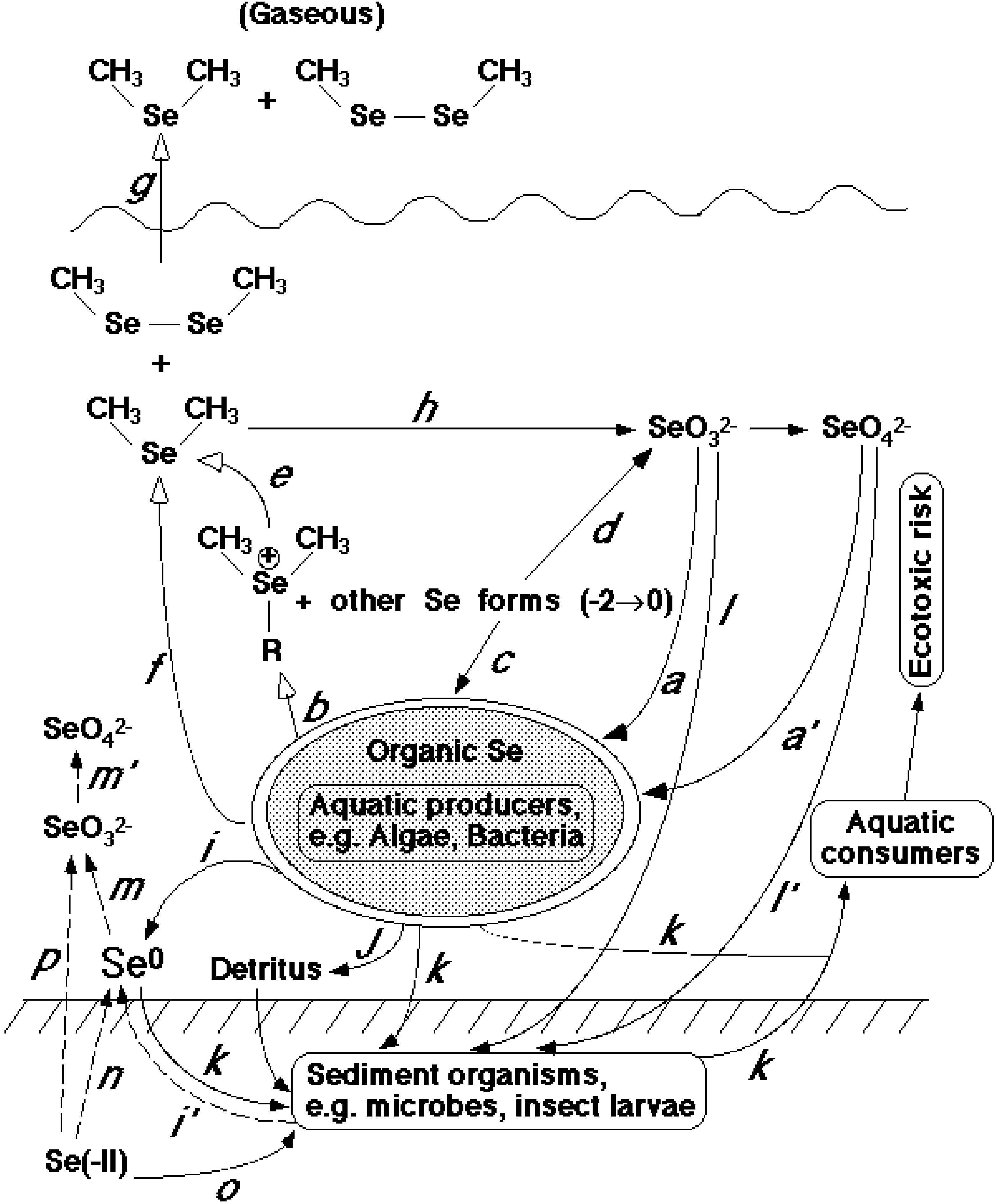

Figure 1. Postulated Selenium Biogeochemical Cycle. This scheme is modified from Cooke & Bruland, 1987. The filled arrows indicate processes that can lead to ecotoxic risk while the open arrows trace the Se volatilization process via which a net loss of Se from the aquatic system can occur.

In both natural and Se-contaminated waters, the dominant forms of dissolved Se are reported to be selenite (+4 form) and/or selenate (+6 form) (e.g. Cooke and Bruland, 1987; Presser et al., 1994). There are also dissolved organoselenium form(s) present in the water column, but the chemical nature of these forms is largely unknown and their concentrations are generally much lower than those of the inorganic Se forms. However, despite the low concentrations, the organoselenium form(s) may still play a very important role in Se ecotoxic effects (e.g. Rosetta and Knight, 1995; Besser et al., 1993). The dissolved selenium oxyanions are primarily taken up by aquatic producers including algae and bacteria (process *a* / a=), and biotransformed into organoselenium form(s) and selenium element (Se^0) (process *i*). These, together with other particle-bound Se sources, constitute the particulate Se of the water column. The biotransformation processes and products in the particulate Se fraction are poorly understood (see Sections II & III). Once accumulated in the aquatic producers, Se can be transferred through various aquatic consumers (e.g. zooplankton, insect larvae, larval fish, bivalves, etc.) into the top predators such as aquatic birds and piscivorous fish (process *k*). Se biomagnification and further transformation can occur during this foodchain transfer process.

It should be noted that the microscopic plantonic organisms including algae, bacteria, protozoan, and zooplankton form a major part of the particulate matters in the water column. These particulate matters, in turn, form the basis for detrital materials which can settle onto the sediment (process *j*) and become the food source for sediment organisms (process *k*). In addition to this Se input into the sediment, waterborne selenite and selenate can be physically adsorbed onto the sediment particles, ingested, absorbed, and transformed by the sediment organisms (process *l* / *l*=). Sediment-bound selenate and selenite can be reduced to insoluble Se^0 by anaerobic microbial activities (process *l*=). This and water column-derived Se^0 can be reduced further to selenide (-2 form) (process *n*) and/or reoxidized to selenite and selenate (process *m* / *m*=) by microorganisms in the sediment and/or in the guts of sediment macroinvertebrates. Selenides can enter the foodchain via absorption into sediment organisms (process *o*) or be oxidized to selenite and selenate (process *p*). Selenium of different oxidation states can be further biotransformed by sediment organisms and transferred up to the foodchain (process *k*). Selenium biotransformation, bioaccumulation, and transfer through both sediment and water column foodweb constitute a major concern for ecotoxic risk in aquatic ecosystems.

In addition to accumulating Se into the biomass, the aquatic producers are the main drivers for the volatilization of Se via the production of methylated selenides including dimethylselenide (DMSe) and dimethyldiselenide (DMDSe) (process *f*). These methylated selenides can be oxidized to selenite (process *h*) or exit the water column into the atmosphere (process *g*). Se volatilization into the atmosphere may represent an important process via which a significant loss of Se occurs in some aquatic systems. Methylated selenides can also be generated from dissolved selenonium precursor(s) (process *e*) released by aquatic producers into the water (process *b*). Moreover, other organoselenium forms can be released into the water by aquatic producers and are reoxidized (process *d*) to selenite and/or reabsorbed by aquatic producers (process *c*).

3. Relationship of the Selenium Biogeochemical Cycle to Stream Dynamics

Given what is known of the complexity of the Se biogeochemcal cycle, water quality objectives that successfully achieve ecosystem protection for a given stream will need to be site-specific, rather than uniform for all environments. The following is a qualitative assessment of the hazard potential of a given stream based on the Se transformations depicted in Figure III-1. Se entry into the food chain via aquatic producers such as algae (process *a*), depends on the stream conditions that stimulate the production of these organisms. For example, algal growth depends on nutrients, temperature, pH, and light (which can be affected by turbidity). Therefore, process *a*, which is important in introducing Se into the food chain, is dependent on stream water characteristics.

Se transformations are affected by redox potentials in the water and in the substrata (processes *1,1', I= and n*). The oxygen concentration is affected by factors such as water depth, flow velocity, flow turbulence and the nature of the stream bed (gravel, rock, sand, silt, or clay). The adsorption of Se on stream bed material is also dependent on the nature of the material. The greatest opportunity for retention of Se in the stream bed is on fine clay and organic materials deposited on the stream bottom. Canton and Van Derveer (1997) and Van Derveer and Canton (1997) proposed and provided some supporting evidence that the Se concentration in the sediment was related to the product of Se concentration in water and the total organic carbon concentration of the sediment. However, this relationship requires further exploration since sediment Se and organic carbon concentrations are highly variable and it is unclear whether they always covary.

All of the above listed factors vary among streams, and season and location within a given stream. Variable stream flow rates also affect the potential of Se to cause ecological damage. For example, very high seasonal flow rates may flush out stream sediments. Based on these factors, the ecological risk of Se in a given stream or other water body will most likely be site-specific. Unfortunately, scientific information is not presently available to quantify the risk under a given set of conditions.

4. Selenium Bioaccumulation and Ecotoxic Effects

Selenium is both an essential nutrient and toxicant, and the margin between nutritional requirement and toxic effects is unusually narrow. Waterborne and sediment Se bioaccumulates readily into the aquatic biota (from primary producers through levels of consumers to top predators) with a typical concentration factor of 1,000 or more (Ohlendorf, 1997; Maier and Knight, 1994). The extent of bioaccumulation depends on the route of exposure (e.g. diet, water, or sediment) and chemical form of Se with a general order of organic forms > selenite > selenate (Besser et al., 1993; Maier and Knight, 1994). In Se-laden environments, chronic toxicity resulting from dietary Se uptake (e.g. through primary production and sediment detritus) and foodchain transfer represents a far greater problem than acute toxicity associated with direct water exposure (Saiki et al., 1993; Canton and Van Derveer, 1997; Maier and Knight, 1994). Chronic Se toxicity is, to an extent, related to waterborne Se concentration and Se bioaccumulation, as shown for the Kesterson Reservoir, other evaporation basins of the San Joaquin Valley (Skorupa and Ohlendorf, 1991) and fly-ash receiving Belews Lake (Lemly, 1985). This relationship is most likely mediated through planktonic uptake of dissolved Se.

However, this relationship is not always applicable to all aquatic environments and cases have been reported where waterborne Se concentrations, Se bioaccumulation, and apparent biological impact did not correlate (e.g. Hamilton, 1997; Hamilton et al., 1997; Van Derveer, 1997; Canton and Van Derveer, 1997; Lemly, 1993; Adams et al., 1997b). Some of these cases are derived from fast-flowing streams with short retention times, as in contrast to the slow-flowing Kesterson reservoir or Belews Lake. In addition, a drastic decrease in Se accumulation in avian eggs was recently observed at a slow-flowing evaporation pond (Rainbow Ranch, Kern County, CA) after a moderate dilution of waterborne Se concentration with agricultural tail water (Anthony Toto, CVRWQCB and Des Hayes, CDWR, personal communication). Consequently, there is a general consensus that complex Se biogeochemistry, particularly biotransformed Se forms in the foodchain, may be the key to chronic Se effects expressed in the top predators such as fish and birds. Since Se biogeochemistry varies with site conditions (e.g. fast-flowing river versus slow-flowing wetlands or ponds), the need for site-specific water quality criteria was urged (Adams et al., 1997a). However, careful and systematic studies that link biological impact to site biogeochemistry will need to be established before such criteria can take hold.

5. Selenium Biotransformations and Ecotoxic Effects

Current knowledge of Se biotransformations has been largely derived from biomedical and nutrition research involving laboratory mammals, heterotrophic microorganisms, and crop plants (e.g. Ganther, 1974; Lewis, 1976; Brown and Shrift, 1982; Doran, 1982). It is clear that Se oxyanions such as selenate and selenite are readily taken up by plants and microorganisms (food sources for animals), and transformed into various organo-selenium forms with selenoamino acids and volatile alkyl selenides among the most commonly occurring forms. It is reasonable to assume that the bioavailability and foodchain transfer potential of alkyl selenides are very low due to their low water solubility and rapid loss from waters. Consequently, the nonvolatile Se form(s) is the key to bioavailability and biotransfer.

The bioavailability of Se in various food sources varies drastically from <5 percent in mushroom to >95 percent in Brasil nut (Thomson, 1997). However, the chemical form(s) of Se that underlie this difference in bioavailability are not known. On the other hand, in feeding studies, selenomethionine was found to be retained in tissues and proteins to a much greater extent than selenite, selenate, or selenocysteine (Thomson, 1997). In addition, dietary selenomethionine gave a similar toxicity profile as that observed for wildlife naturally exposed to Se (e.g. Heinz et al., 1988 & 1989). Moreover, proteinaceous selenomethionine was recently found to be the major form present in bird eggs collected from a Se-laden drainage system of the San Joaquin Valley (Fan, Skorupa, and Higashi, unpublished result). Whether selenomethionine is a key form that lead to Se bioaccumulation and toxicity in aquatic ecosystems will need to be further investigated.

Although microalgae are often the dominant primary producers in many aquatic environments including the San Joaquin River and most evaporation basins of the San Joaquin Valley, only a few biotransformation studies have been reported for microalgae, all with a focus on Se incorporation into proteins (e.g. Wrench, 1978; Bottino et al., 1984; Price and Harrison, 1988). There has been some hint about the importance of microalgae in the production of alkyl selenides and their selenonium precursors in ocean and inland waters including the Kesterson Reservoir and San Joaquin River

(Cooke and Bruland, 1987; Amouroux and Donard, 1996).

6. Selenium Interactions with Other Contaminants

It has long been known that Se interacts with other trace elements such as Hg, Cd, and As in terms of their biological effects (e.g. Ganther, 1974; Naddy et al., 1995). In many cases, Se counteracts the effect of other toxic elements. For example, Se prevented the leg paralysis of adult mallards caused by Hg poisoning (Heinz and Hoffman, 1996). However, in the same report, Se was found to greatly enhance Hg toxicity on hatching success and survival of mallard ducklings. The molecular mechanism(s) underlying these interactions are unknown but Se forms are considered to be the key to this understanding.

7. Present Knowledge of the San Joaquin River Ecosystem

The bioaccumulation and impact of Se from agricultural drainwaters on wildlife have been documented in the Kesterson Reservoir and San Luis Drain system. For example, unusually high levels of Se body burden in the order of hundreds of μg/g wt were found in organisms throughout the foodweb. These included algae, macrophytes, benthic invertebrates (e.g. chironomids), fish (e.g. mosquitofish) (Saiki and Lowe, 1987), and waterfowls (e.g. stilts, avocets) (Ohlendorf et al., 1990). The elevated Se burden was in turn linked with severe deformities and/or reproductive failures observed for waterfowls at Kesterson (e.g. Ohlendorf et al., 1986a&B). In addition, impaired reproduction was correlated with the Se exposure for western mosquitofish (*Gambusia affinis*) from the San Luis Drain (Saiki and Ogle, 1995).

From both laboratory exposure experiments and field survey, it is evident that dietary uptake and foodchain transfer represented a major route via which Se was bioaccumulated in wildlife (Ohlendorf et al., 1993; Maier and Knight, 1994). It is also clear that organic-rich sediments resulting from high primary production and waterborne Se concentrations of the order of hundreds of μg/L are a key to wildlife toxicity observed at these sites.

Relative to the Kesterson Reservoir, less Se body burden data are available from the San Joaquin River and its tributaries, and it is more difficult to relate the Se body burden to waterborne Se concentrations, since the temporal and spatial resolution was insufficient plus the river system is much more dynamic with open demography (i.e. organisms can migrate in and out of a contaminated river segment). Moreover, the relationship between Se body burden and adverse effects is more difficult to define and predict than the Kesterson case for the following reasons: the effects are expected to be more subtle, impacted organisms are not confined to the contaminated sector, and no defined laboratory studies have been conducted on indigenous organisms under appropriate exposure regime and conditions. Such studies are critically needed since Se burden in the river foodchain has been generally lower than that reported for the Kesterson case (cf. Saiki et al., 1993 and Saiki and Lowe, 1987; Leland and Scudder, 1990).

Although short-term acute toxicity is unlikely to occur due to tile drainage discharge into the river, there are lines of evidence that long-term impact will need to be addressed. First, waterborne, sediment, and detrital Se concentrations are elevated near the tile drainage-influenced areas relative to other parts of the river system (see CVRWQCB water quality data; Leland and Scudder, 1990; Saiki et al., 1993). The detrital Se, in particular, has been recognized as a major dietary source for benthic macroinvertebrates which in turn may be main diets for fish and birds (Saiki et al., 1993). Second, benthic bivalves (e.g. *Corbicula* sp. and chironomid larvae) and fishes (e.g. common carp, mosquitofish, and juvenile striped bass) (Johns et al., 1988; Saiki et al., 1992 & 1993; Saiki and Palawski, 1990; Leland and Scudder, 1990) exhibited elevated Se body burden in drainage service areas of the river. Third, Se burden in the developing ovaries of long-lived organisms such as the white sturgeon was up to 72 μg/g dry wt. Se burden in the sturgeon eggs was also elevated (up to 29 μg/g), most of which resided in the yolk proteins (Kroll and Doroshov, 1991). Although the impact of these elevated Se burdens on sturgeon reproduction is unclear, it is possible that they may lead to reproductive failures since preliminary work by Fan and Higashi indicated that the majority of Se in deformed bird embryos resided in the protein fraction (see also above).

The unusually high Se burden in the sturgeon brings up the issue as to how Se is transferred to this predator, and how drainage discharge can affect this process. In the short term, Se input from the drainage is not expected to have a significant effect on Se bioaccumulation into white sturgeon, since its main food source has shifted to the Asian clam (*Potamocorbula amurensis*) which is of the Delta/Bay origin, according to the surveys of 1965-67 (McKechnie and Fenner, 1971) and 1990 (SWRCB, 1991). However, in the long term, it is unclear how much the drainage source would contribute to the Se load in white sturgeon. This aspect should be carefully monitored because of the major shift of benthic macroinvertebrate community to the Asian clam in the Bay/Delta since 1986 (Carlton et al., 1990) and potential invasion into the San Joaquin River. This clam accumulates Se to unusually high levels (Brown and Luoma, 1995), presumably due to its efficient filtration capability.

8. Selenium Biotransformations in the Food Chain of the San Joaquin Valley

Few studies have been conducted to investigate Se biotransformations by aquatic organisms inhabiting the San Joaquin Valley. There has been no information available on important foodchain organisms such as benthic macroinvertebrates. Regarding primary producers, it was only recently demonstrated that several species of microalgae isolated from the drainage waters of the Valley actively transformed selenium oxyanions into alkylselenides, selenonium ions, and proteinaceous selenomethionine (Fan et al., 1997; Fan and Higashi, 1998; Fan et al., 1998a&b). Se volatilization accounted for 60-70 percent of the total Se loss from the medium in laboratory studies of one of these species (a filamentous cyanophyte) (Fan et al., 1998b). However, a significant amount of Se was also bioconcentrated in this alga, particularly in proteins where selenomethionine was the dominant form. There was also a major difference in Se allocation into proteins among different algal species (Fan et al., 1998a), which suggests a very different foodchain transfer potential since diet with a higher protein level led to a higher Se content in fish tissues (Riedel et al., 1997). In addition, the alga (filamentous cyanophyte) with a higher Se content in proteins demonstrated a lower tolerance to waterborne Se exposure than one (*Chlorella* sp.) with a lower proteinaceous Se content (Fan et al., 1998a).

9. Assimilative Capacity of Biomagnifying Trace Elements (Manucher, is this section still in the draft? If it is, then the statement on Se volatilization may need to be modified.)

The general relationships between load, water volume, concentration, and assimilative capacity apply for constituents, such as most dissolved salts in water, that do not undergo major chemical transformation. A good relationship between concentration and biological impact usually exists for constituents which do not become biomagnified in the food chain. As explained in section III, selenium undergoes both chemical transformation and biomagnification. Therefore, water quality criteria appropriate for salts are inadequate for selenium, as account must be made for transformations and biomagnification in calculating assimilative capacity. As a simple hypothetical example, assume conditions in water body A cause more selenium volatilization than water body B. Since volatilization removes selenium from the food chain, water body A could receive more selenium than water body B with the same biologic impact. In other words, the assimilative capacity of water body A is greater than water body B, assuming everything else is equal.

Many field cases could be cited illustrating selenium transformations in aquatic systems. One example is estimated selenium loss in agricultural drainage flowing through Grassland area channels between 1986 and 1994 (Nigel Quinn, personal communication). There was an average reduction in selenium in the water as it flowed through the channels of about 24 percent. The selenium may have been deposited in channel sediment or entered the shallow groundwater through seepage, although the fate of the selenium removed from the water and the biological impact are not known for certain. What is known is that selenium reacts differently than other elements in an aquatic system. Furthermore, it is known that the amount of selenium discharged into the SJR was reduced in drainage flowing through the system of channels and wetlands, thereby reducing the potential negative biologic impact to the SJR by some unknown amount. A quantitative comparison of negative biologic impacts to the Grassland ecosystem with the beneficial biologic effects to the SJR must be interpreted cautiously because of incomplete scientific understanding of the various selenium biotransformations and biomagnification.

III.C. Boron

Boron freshwater chemistry, characteristic of most natural waters in the SJR Basin, involves two species, $B(OH)_3$ and $B(OH)_4^-$. Soils have a large capacity for boron adsorption as both species may be adsorbed on the surfaces of various clay minerals, hydroxy oxides of Al, Fe, and Mg, and organic matter (Keren and Bingham, 1985). Crop toxicity may occur if the soil adsorption capacity is exceeded, resulting in an increase in boron availability and uptake by plants (Eisler 1990; Gupta et al. 1985).

As pH increases to about 9.5, $B(OH)_4^-$ concentration increases rapidly as does B adsorption. This is consistent with reduced plant uptake of boron which occurs with increasing soil alkalinity (Butterwick et al., 1989). Further increases in pH result in decreased boron adsorption due to competition of OH^- for the adsorption sites. Boron adsorption also increases with salinity. Adsorption decreases plant uptake and facilitates boron transport in aquatic systems.

The behavior of boron in natural waters and soils is complicated by the presence of other constituents. Interactions with commonly dissolved salts and minor elements can sometimes make the relationship between laboratory and field results confusing. For example, the boron tolerance of many plant species may be enhanced by increasing levels of soil salinity (Ferreyra, et al, 1997).

Boron Impacts by Water Use

Selected research information is summarized below to provide insights into what is known about boron impacts on beneficial water uses for crops, human health, cattle, waterfowl, fish and amphibians, and other aquatic life.

Crop Use

Boron is essential for the growth of higher plants in relatively small quantities, but is toxic in slightly greater amounts. Leaves normally contain about 40 to 100 mg/kg (ppm). Concentrations may exceed 700 B 1000 mg/kg where boron toxicity occurs.

In Table III-1, crops are grouped according to tolerance to boron; concentrations in soil water extract at which plant damage occurs are shown in parentheses. Most of these concentrations were obtained during greenhouse experiments conducted by Eaton (1944), and are the lowest level of boron in culture solutions that caused visual damage. Maas (1990) commented on the boron tolerance data of Eaton (1944): " Although useful, they cannot be fitted to any reliable growth response function for most crops.@ Work conducted by Francois (1984,1986, 1988, 1989, 1991, and 1992) with vegetable crops indicated that

Table III-1 RELATIVE BORON TOLERANCE OF AGRICULTURAL CROPS
(Maas 1990; Francois 1991 and 1992)

Very Sensitive (<0.5 mg/L)

Lemon	*Citrus*
Broccoli	*Brassica Olerace abotrytis*
Blackberry	*Rubus spp.*

Sensitive (0.5-0.75 mg/L)

Avocado	*Persea americana*
Grapefruit	*Citrus X paradisi*
Orange	*Citrus sinensis*
Apricot	*Prunus armeniaca*
Peach	*Prunus persica*
Plum	*Prunus domestica*
Persimmon	*Diospyros khaki*
Fig, kadota	*ficus carica*
Grape	*Vitis vinifera*
Walnut	*Juglans regia*
Pecan	*Carya illinoiensis*
Turnip	*Brassica rapa*

Sensitive (0.75-1.0 mg/L)

Sweet potato	*Ipomoea batatas*
Wheat	*Triticum eastivum*
Sunflower	*Helianthus annuus*
Bean, mung	*Vigna radiata*
Sesame	*Sesamum indicum*
Lupine	*Lupinus hartwegii*
Strawberry	*Fragaria spp*
Artichoke, Jerusalem	*Helianthus tuberosus*
Bean, kidney	*Phaseolus vulgaris*
Bean, lima	*Phaseolus lunatus*
Peanut	Arachis hypogaea

Moderately Sensitive (1.0 - 2.0 mg/L)

Bean, snap	*P. vulgaris*
Pepper, red	*Capsicum annuum*
Pea	*Pisum sativa*
Carrot	*Daucus Carota*
Radish	*Raphanus sativus*
Potato	*Solanum tuberosum*
Cucumber	*Cucumis sativus*
Lettuce	*Lactuca sativa*
Cherry	*Prunus avium*

Moderately Tolerant (2.0-4.0 mg/L)

Cowpea	*Vigna unguiculata (L.)*
Walp	
Cabbage	*Brassica oleracea capitata*
Bluegrass,	*Poa prtensis* Kentucky
Oats	*Avena sativa*
Maize/corn	*Zea mays*
Artichoke	*Cynara scolymus*
Tobacco	*Nicotiana tabacum*
Mustard	*Brassica juncea*
Clover,sweet	*Melilotus indica*
Squash	*Cucurbita pepo*
Muskmelon	*Cucumis melo*

Tolerant (4.0-6.0 mg/L)

Tomato	*Lycopersicon*
Alfalfa	*Medicagom sativa*
Vetch, purple	*Vicia benghalensis*
Parsley	*Petroselinum crispum*
Beet, red	*Beta vulgaris*
Sugar beet	*Beta vulgaris*
Cauliflower	*B. Oleracea botrytis*
Garlic	*Allium sativum*

Very Tolerant (6.0-15.0 mg/L)

Onion	*Allium cepa*
Sorghum	*Sorghum bicolor*
Cotton	*Gossypium hirsutum*
Asparagus	*Asparagus officinalis*
Celery	*Apium graveolens*

Toxicity levels based on Eaton's work did not correlate well with the effect of boron on yields of cowpea, onion, garlic, lettuce, and celery. With the exception of lettuce, these crops were reclassified into a more tolerant category.

Unfortunately, for tree and vine crops, little research data exists on the linkage between Eaton's visual damage and crop yields. For grapes, Christensen and Ayers (1974) reported that boron levels in irrigation water above 1 mg/L in the San Joaquin Valley caused some yield loss. Based on a survey of farm advisors and consultants in California conducted by Oster for the purpose of this report, tree crops in the very sensitive to sensitive categories in Table III-1 are likely conservatively classified (i.e., classified at a more boron sensitive level that may actually be the case). The beneficial effects of rainfall, which commonly occurs in many of the tree and vine cropping areas of California, would be one reason that the effects of boron on yields of tree crops and grapes are less than those expected based on Eaton's work. A lack of correlation between visual B symptoms and boron effects on yield is another.

Human Use

Klasing and Pilch (1988) stated that some human and animal studies indicated adverse male reproductive effects from "very high levels" of dietary boron (e.g. 0.3 mg/kg of body weight for rats exposed over six months). However, they concluded that acute and/or chronic dose-response, which was shown to cause such effects, was conflicting. They stated that additional studies were particularly needed to determine chronic dose-response effects.

Citing Nielsen (1994) that boron could be an essential trace element for humans, Murry (1995) states that there is insufficient data to establish an essential nutritional need for boron. Using a relative source concept, Murry (1995) did a human health risk assessment of boron in drinking water. An acceptable daily intake of 18 mg boron/day was obtained for a women of child bearing age with an average weight of 60 kilograms. This was based on no observed adverse effect level (NOAEL) of 9.6 mg boron/kg/day for developmental toxicity in rats, and division of this number by 32 to account for intraspecies and interspecies variation. Subtracting an average dietary intake of 1.5 mg boron/day from food, resulted in an acceptable drinking water uptake of 16.5 mg boron/day. Based on a drinking water consumption of two liters/day, a person could drink water containing up to 8.25 mg/liter boron. Murry's conclusion from his risk assessment is that consuming water with up to 4 mg/L boron per day would not pose any developmental, reproductive, or other health risk to the public. However, based on a two-year dog study for testicular atrophy and spermatogenic arrest, USEPA revised their NOAEL boron reference dose for chronic oral exposure in their Integrated Risk Information Systems (IRIS) database in June 1995. The NOAEL for this study was 8.8 mg/kg/day and the Reference Dose (RfD) was 0.09 mg/kg/day. The resulting lifetime health advisory level for boron was 0.63 mg/L. In addition, a state action level of 1.0 mg/L was published by the California Department of Health Services (Marshack, 1998).

Cattle Use

Ayers and Westcot (1985) suggest a 5 mg/L guideline given for cattle use based on guidelines prepared by the National Academy of Sciences (1973), which were purposely set to provide a wide safety margin. Nielsen (1986) concluded that livestock showed signs of adverse effects from boron in drinking water at concentrations over 150 mg/L. Butterwick et al. (1989) summarized boric acid toxicity to cattle from two drinking water studies as follows:

$ Swelling and irritation of legs, lethargy and diarrhea occurred from 30 days exposure to boron acid at concentrations of 150 to 300 mg/L boron.

$ No signs of toxicosis were observed from exposure to 120 mg/L boron for 10 days.

Waterfowl Use

Smith and Anders (1989) reported exposure to 1,000 mg/kg dietary boron in breeding mallards caused an increase in embryo and hatching mortality. Embryo growth reduction was recorded when hens were exposed to 300 and 1,000 mg/kg dietary boron. Hatchling weight gain was reduced at concentrations as low as 30 mg/kg dietary boron. Hoffman et al. (1990) found a 10 percent mortality in one-day-old mallard ducklings exposed to concentrations of 1,600 mg/kg boron and growth reductions at concentrations of 100, 400 and 1,600 mg/kg of boron.

According to Perry and Suffet (1994), only four laboratory studies have addressed boron bioaccumulation in waterfowl tissues. Dietary boron at 1,600 mg/kg produced concentrations in brain and liver tissues that were 25 and 29 times greater than the concentrations found in the corresponding tissues of control animals. Significant boron bioaccumulation in liver and brain tissue was reported when dietary boron concentrations were between 100 and 1,600 mg/kg for 10 weeks.

Eisler (1990) reviewed the literature and noted that dietary concentrations of 300 to 400 mg boron/kg in feed (fresh weight) affected mallard growth, behavior and brain chemistry. Dietary boron levels of 100 mg/kg fresh weight reduced growth of female mallard ducklings (Hoffman et al. 1990). Dietary boron as low as 30 mg/kg fresh weight fed to mallard adults affected offspring growth rates (Smith and Anders 1989).

Fish and Amphibian Use

According to Birge and Black (1977) who examined boron toxicity four days after hatching, the embryo stage aquatic concentrations in mg/L boron at which 1 percent mortality (LC_1) and 50 percent mortality (LC_{50}) follow:

Aquatic Species	LC_1	LC_{50}
trout	0. 001 to 0.1 mg/L	27 to 100 mg/L
goldfish	0.2 to 1.4	46 to 75
catfish	0.2 to 5.5	22 to 155
amphibians	3 to 25	47 to 145

LC_{50} values were significantly higher than LC_1 values for all species. Birge and Black (1977) compared their results with other published data and concluded that boron compounds were more toxic to developmental and early posthatched stages than to adult fish. They also concluded from an analysis of variance that boric acid was significantly more toxic than borax ($Na_2B_4O_7X14H_2O$) to fish embryos. Hardness of water did not exert a statistically significant effect on boron toxicity, but a trend showed toxicity to embryonic stages generally was greater in hard water. In general, they state that boron concentrations of 100 to 300 mg/L were lethal for all species tested.

Coho salmon (Oncorhyncus kisutch) *under*-yearlings, as reported by Thompson et al. (1976), when exposed for 12 to 23 days showed an LC_{50} of 113 mg/L of boron. Chinook salmon (Oncorhyncus tshawytscha) as "swim-ups" and advanced fry had a four-day LC_{50}- of 725 mg/L as reported by Hamilton and Buhl (1990). They also reported coho salmon as "swim-ups" and advanced fry had a four-day LC_{50} of 447 mg/L boron.

Hamilton and Wiedmeyer (1990) found no boron detected in fish when exposed to concentrations as high as 6 mg/L. Using water from a Westlands Water District sump with boron concentrations ranging from 44 to 53 mg/L, Saiki, et al. (1992) studied the toxicity of San Joaquin Valley water to juvenile chinook salmon and striped bass (Morone saxatilis). They found chinook salmon and striped bass exposed to the drain water had boron concentrations as high as 200 ug/g on a dry weight basis. They concluded that elevated concentrations of trace elements (especially boron and selenium) may have contributed to the toxicity of the drainage water, but the extent was not clearly defined. Saiki et al (1992) noted that Hamilton and Buhl (1990) found boron as relatively non-toxic to swim-up and advanced fry of chinook salmon. They also stated that reports by USEPA (1986) and Eisler (1989) suggest that boron did not contribute greatly toward overall toxicity of the drain water.

Results from laboratory and field studies suggest that boron bioaccumulation occurs in fish, but significant bioconcentration does not (Perry and Suffet, 1993; Hamilton and Wiedmeyer, 1990; Saiki and Maya, 1988, and Ohlendorf et al., 1986). Butterwick et al. (1989) after summarizing toxicity data for amphibians, invertebrates, algae and other aquatic life stated that no evidence has been found that aquatic organisms bioaccumulate boron.

How does boron toxicity hazards compare to other potentially toxic elements? Hamilton (1995) conducted acute toxicity tests on three life stages of Colorado squawfish (*Ptychocheilus lucius*), razorback sucker (*Xyrauchen lexanus*), and bonytail (*Gila elegans*) in a reconstituted water quality that simulated the Green River of Utah. He conducted tests with boron, lithium, selenate, selenite, uranium, vanadium, and zinc. Boron was ranked as the least toxic of these chemicals to three life stages (swim-up and two juvenile) of these fish species. Acute toxicity for boron at the 96-hour LC_{50} ranged from 100 to 527 mg/L.

Freshwater Plants

Stanley (1974), as cited in Butterwick et al., (1989), observed that a concentration of 40.3 mg/L boron lead to a 50 percent inhibition of root growth in *Myriophyllum spicatum* after 32 days of treatment. For duckweed (*Lema minor*), three days of exposure to 200 mg/L resulted in signs of toxicity (Frick, 1985), and normal growth occurred at 10 to 20 mg/L. Glandon and McNabb (1978) observed no adverse effects on duckweed growth at 0.0 1, 0.11, and 1.01 mg/L boron.

Perry and Suffet (1994) reported that Bowen and Gauch (1966) observed a reduction in growth rate for green algae (*Chlorelia vulgaris*) at a boron concentration of 50 mg/L and a reduction in *C. prothicoides* and *C. emersanii* growth at a boron concentration of 100 mg/L. Neither the number and weight of *C. vulgaris* cells were stimulated nor inhibited by 0.5 mg and 10 mg/L boron (McBride, et al., 1971). Boron does not seem to be required by green algae for growth (Gerloff, 1968).

For blue green-algae (*Anacysis nidulans*), Martinez, et al., (1986) reported that boric acid concentrations of 10, 25, and 50 mg/L did not affect the growth rate or chlorophyll and protein content over a 96-hour exposure. However, 75 and 100 mg/L resulted in a decrease in growth rate and chlorophyll content. At 50, 75, and 100 mg/L of boron, they reported a reduction in growth and a drop in proteins, chlorophyll, and phycobiliproteins in the blue-green algae species, Anabaena PCC 7119. Phytoplankton can tolerate up to 10 mg/L inorganic boron in the absence of other stresses (Antia and Cheng, 1975; Eisler, 1990).

Bringmann (1978) noted that cell replication in fresh water protozoan (*Entosiphon sulcalum*) was reduced by 5 percent when exposed to 1 mg/L boron for 3 days. Kapu and Schaeffer (1991) examined behavior responses in the flatworm planarian (*Dugesia dorotocephala*) after exposure to various concentrations of metals including boron at 1 to 60 minute intervals. Effects on behavior -- mostly restlessness, hyperkinesia, spiraling, and reed/nose twist -- were observed at 1 mg/L boron.

Invertebrates

According to Eisler (1990), no observable effects were seen on water flea (*Daphnia magna*) at a boron (as boric acid) concentration of 13.6 mg/L. A few studies focused on acute and chronic, lethal and sublethal effects of boron for water flea.
No observable effect concentration (NOEC) and lowest observable effect concentration

(LOEC) values were calculated at 6 and 13 mg/L boron (Butterwick et al. 1989). Studies by Lewis and Valentine (1981) and Gersich (1984) reported 48-hour LC_{50} values of 226 and 133 mg/L and 21-day LC_{50} values of 53.2 and 52.2 mg/L, respectively.

Maier and Knight (1991) found lethal and sublethal toxicity for water flea (*Daphnia magna*) and benthic invertebrate midge (*Chironomus decorum*) when exposed to tetraborate. The 48-hour LC_{50} for the water flea was 141 mg/L. The 48-hour LC_{50} for C *decorum* was 1,376 mg/L. A 48-hour exposure to a boron concentration of 20 mg/L resulted in a significant decrease in midge larval growth rate. For the most sensitive species of mosquito larvae, preliminary investigations by USEPA (1975) showed a 48-hour LC_{50} boron concentration of 700, 524, 1,748 and 2,797 mg/L for four stages of development.

Water Quality Standards and Research Needs

USEPA (1986) has an agricultural water quality criterion for boron of 0.75 mg/L to protect sensitive crops during long-term irrigation (Marshack 1998). Ayers and Westcot (1985) recommended a concentration of 0.7 mg/L boron in water that would require no restriction for agricultural use. These agricultural criteria, which are conservative for almost all crops, particularly where rainfall meets some of the crop water requirement, are likely also conservative for other beneficial uses.

Livestock drinking water uses do not appear to be particularly sensitive to boron. Livestock are tolerant to boron in drinking water with 5 mg/L being the guideline given by Ayers and Westcot (1985).

No California or federal drinking water standards have been established for boron. However, as a reference, the California Department of Health Services has set a State Action Level of 1 mg/L. USEPA Integrated Risk Information System (IRIS) has a Reference Dose of 0.63 mg/L for noncarcinogenic effects. Klasing and Pilch (1988) stated that additional studies were need to find chronic dose-response health effects of boron.

Boron standards have not been set for the protection of aquatic life. For toxic chemicals, the four basic methods that have identified as potential approaches for developing criteria for aquatic life are: (1) USEPA's National Guidelines Tier I method, (2) Tier I method as modified by the California Department of Fish and Game, (3) the USEPA's Tier II (Great Lakes) method, and (4) modified SWRCB Ocean Plan method. Each of these methods increases in accuracy with more data.

A technical committee looked into setting water quality criteria (SWRCB 1988) for boron but concluded that aquatic toxicity data for boron was limited. They suggested a criterion of 0.55 mg/l based on the most conservative, modified, ocean plan method. However, their number was based on an unusually low LC_{10} of 1.02 mg/L for rainbow trout in Birge and Black (1981), as well as concentrations of 2.0 mg/L for a water plant (*Elodea canadensis*), and 13.0 mg/L for a water flea (*Daphia magna*). Similarly, a University of California Committee of Consultants (1988) evaluated SJR water quality objectives for boron (along with selenium and molybdenum) and recommended developing a larger database on boron toxicity for aquatic plants, which are likely more sensitive to boron than animals.

Perry and Suffet (1994) analyzed data requirements for a boron water quality criterion in aquatic systems, and summarized the literature by stating that lethal effects of boron are apparent at concentrations that are often at least one order of magnitude higher than concentrations at which sublethal effects were observed. They recommended chronic lethal and sublethal boron toxicity tests on freshwater aquatic plants, aquatic invertebrates, fish, amphibians, and waterfowl living in the San Joaquin Valley and believe that a better data base needs to be developed before final objectives for boron can be set within the SJR system.

Saiki (1998) states that existing information on the toxic effects of boron to aquatic organisms is too sparse to warrant anything more than interim water quality objectives for aquatic organisms in the San Joaquin Basin. He states that only a few studies have examined sublethal effects of long-term exposure to dissolved boron and even fewer studies have examined the effects of dietary exposure. He states that more studies are needed before objectives can be set that can confidently protect fish and wildlife resources.

In general, boron water concentrations resulting in lethal effects to aquatic life are at such elevated concentrations that are not typically found in natural freshwater systems, including the SJR system. Toxicity based criteria derived from bioassays measuring mortality as the endpoint may not be sensitive enough to prevent sublethal chronic effects. Short exposure periods used in acute tests (96 hours) do not reflect conditions under which organisms exist in the environment. Chronic exposures at relatively low concentrations are more reflective of the natural environment. There is a lack of sublethal boron toxicity data for aquatic species that inhabit the SJR ecosystem. Overall, more data on toxicity to fish and other aquatic species would be needed to substantiate a numerical criterion for boron based on aquatic life.

Conclusions

In summary, the relative boron tolerance data in Table III-1 are likely conservative for many of the crops. Also, application of the data in Table III-1 to growing conditions in California requires consideration of the beneficial impacts of rainfall on the levels of boron in the soil. Rainfall can dilute boron concentration in the soil water.

Consuming up to 4 mg/L boron per day in drinking water would not be expected to pose any developmental, reproductive, or other health risk to the public. A guideline of 5 mg/L is recommended for livestock which show signs of adverse effects from boron in drinking water at concentrations over 150 mg/L.

Aquatic birds, fish, and invertebrates can be adversely impacted by elevated boron concentrations, but in general, boron water concentrations resulting in lethal effects to aquatic life are at elevated concentrations that are not typically found in natural freshwater systems, including the SJR system. Toxicity based criteria derived from bioassays measuring mortality as the endpoint, may not be sensitive enough to prevent sublethal chronic effects. Short exposure periods used in acute tests (96 hours) do not reflect conditions under which organisms exist in the environment. Chronic exposures at relatively low concentrations are more reflective of the natural environment. There is a lack of sublethal boron toxicity data for aquatic species that inhabit the SJR ecosystem. Overall, more data on toxicity to fish and other aquatic species would be needed to substantiate a numeric criterion based on aquatic life for boron.

III.D. Molybdenum

Molybdenum (Mo) is one of 15 trace elements essential for plant growth with its role primarily in enzymatic activation for nitrogen metabolism. As with many other essential trace elements in biological systems, Mo is required in small concentrations and may be toxic in excess. Molybdate (MoO_4^{-2}) is the principal species that occurs in natural environments (USEPA, 1979; Troeh and Thompson, 1993). Concentrations of Mo of 1 ug/L are common in surface and groundwater, with concentrations greater than 10 to 20 ug/L usually associated with human activities such as mining and industry. Soils irrigated with water high in Mo show greater soil Mo availability and therefore have greater potential for plant uptake of Mo (Vleck, 1976).

In soils, Mo is usually a component of organic matter and minerals, or adsorbed on positively charged exchange sites; its solubility increases with increasing pH (Troeh and Thompson, 1993). The chemistry of molybdenum resembles that of phosphorus with relatively high plant availability when the pH exceeds about 8.5 and low availability at a pH below 6.5.

Soil Mo levels in the United States generally exhibit a marked increase east to west across the continent. In the West, areas with high Mo are localized. In particular, seepage areas at the terminus of wet, narrow floodplains or alluvial fans of small streams can have elevated levels of Mo, indicating that Mo moves with water (Burau and McLean, 1979). Toxic levels of Mo in forage are consistently associated with forage grown on poorly drained western soils.

Molybdenum Impacts by Water Use

A review of water quality concerns with respect to Mo has been done by Chilcott (1998).

Cattle Use

A main concern for Mo is its ability to bioaccumulate in certain plant species, notably legumes, which in turn can cause molybdenosis and infertility in cattle. Symptoms of molybdenosis have been known to occur in ruminants (particularly cattle) grazing on forage containing concentrations of Mo above 10 ppm. Higher levels of Mo in forage (20 - 100 ppm) may be tolerated depending on the copper (Cu) to Mo ratio. For cattle, a Cu:Mo ratio >2 seems to prevent symptoms of molybdenosis; however, molybdenosis has been observed in young cattle in California when the forage contained 4.7 ppm Cu and 1.8 ppm Mo (Fisher, 1978). Concentration levels for the onset of molybdenosis are not absolute as molybdenum interacts with a variety of elements, such as copper and sulfate, and with environmental factors such as soil acidity, which affect the availability and toxicity of Mo.

Mo levels as low as 5 ppm in forage delayed first oestrus in cattle by at least 6 weeks and the pregnancy rate for Mo-treated animals was 30 percent, significantly lower than in the control (Phillippo, et al. 1985). Although copper deficiency symptoms can be easily treated by injections of Cu compounds, fertility symptoms appear to be related directly to the Mo concentration and are not readily correctable.

Human Use

High concentrations of Mo are not currently considered a potential human health hazard as the worst human-related toxicity symptom has been a gout-like disease.

Freshwater Aquatic Life Use

Aquatic organisms are relatively resistant to Mo. In freshwater laboratory studies, some species of aquatic algae and invertebrates significantly bioconcentrated Mo in ambient water concentrations ranging from 0.0005 to 3300 ug/L, yet no effect was observed (Short et al. 1971; Steeg et al., 1986). Freshwater invertebrates and fish appear to be very resistant to Mo; LC_{50} values at 48 and 96 hours (Table III-1) range from 70 mg/L for fathead minnows to about 3620 mg/L for amphipods. One exception is newly fertilized eggs of rainbow trout (fertilization through 4 days post-hatch) with an LC_{50} level about 0.8 mg/L after exposure for 28 days.

Few studies have compared Mo concentrations in fish tissues to ambient concentrations. There are no studies on wild birds, and the current data on wild mammals are inadequate.

Water Quality Guidelines and Objectives

Only limited guidelines exist for water quality concentrations. Even though Mo is not a human carcinogen, a USEPA guideline of 35 ug/L exists for potential human health concerns. For animals, a guideline of 10 ug/L exists for forage irrigation, whereas for drinking water it is 50 ug/L. No objectives have been set for the protection of aquatic life. The current water quality objectives for the San Joaquin River Basin are given in Table IV-2.

The Mo objectives for the SJR could cause concern for cattle use were it not for several factors which reduce the potential hazard existing levels pose to animal fertility:

1. Rainfall amounts in addition to irrigation are a significant source of water to forage crops grown north of Mendota.

2. Forage crops irrigated with SJR water will not likely be the sole source of forage for cattle.

3. Mo is partially leachable in soils with pH levels higher than 7.0.

4. Sulfate reduces Mo uptake.

CURRENT SITUATION

Grassland Watershed

Soil conditions in the Grassland Watershed are consistent with expected high Mo levels. Clawson (1973) found molybdenosis in both Merced and Fresno Counties in locations surrounding the Grassland Watershed. Consequently, the CVRWQCB began monitoring for Mo at key sites listed in Table III-2, and found that agricultural subsurface drainage sumps in the 90,000-acre DPA contained the highest concentrations of Mo, averaging 50 ug/L with many concentrations exceeding 100 ug/L (Chilcott et al., 1988). Waters that enter SGWD, Camp 13 Slough, and Agatha Canal have lower Mo levels (Table III-2), with maximum concentrations not exceeding 34 ug/L during the period of record, and mean and median concentrations remaining near or below the 10 ug/L water quality objective for forage and pasture irrigation. The lower concentration may be due to a number of factors, including mixture of subsurface drainage with both agricultural tail water and surface water. The Cu:Mo ratio was two or greater (Table III-2) for waters entering SGWD, indicating less chance of a problem of animal toxicity from forage.

Mo concentrations were variable for water bodies exiting the SGWD, (Table III-2). Mo concentrations for both Salt Slough and Mud Slough (north) were compared to the mean-monthly objective of 19 ug/L. Although only a single sample from Salt Slough exceeded this objective, many grab samples from Mud Slough (north) exceeded the objective. The maximum objective of 50 ug/L was not violated in either Slough. The mean-monthly objective appears to have been violated only when no subsurface drainage or wetland releases were entering Mud Slough (north), indicating that normal seepage or background flow in the Slough contains highly elevated levels of Mo (Westcot et al., 1992).

Prior to the implementation of the GBP in 1996, Mo concentrations in the SLD were highly elevated with average values exceeding 90 ug/L (Table III-2). The elevated Mo levels were the result of evapoconcentration of groundwater seeped into the SLD over time. Concentrations decreased significantly when the GBP began operation, averaging 27 ug/L and ranging from 22 to 35 ug/L during water year 1997. After operation, the drainage water is flowing in the SLD and no stagnant condition exists which would result in evapoconcentration of Mo.

Lower San Joaquin River

Mo concentrations are consistently higher at sites upstream of discharges from the Grassland Watershed (Table III-2). Lander Avenue is the site for the CVRWQCB=s lower SJR background monitoring. Because most of the SJR flow is diverted into the Friant-Kern Canal, sections of the SJR upstream of Mendota Pool are usually dry except during periods of wet weather or major snow melt. During the irrigation season, flow at Lander Avenue downstream of Mendota Pool consists largely of groundwater accretion. Consequently, Mo concentrations are consistently higher than at sites downstream of the Mud and Salt Slough confluences. Only at the Lander Avenue site have Mo concentrations exceeded the 19 ug/L mean-monthly objective, and the 50 ug/L maximum objective on occasion. The objectives were exceeded during periods of extremely low flow, prior to winter storms. However, these exceedances are background concentrations with no known source except groundwater accretion.

At the Hills Ferry site, Mo concentrations did not exceed 19 ug/L, and normally remained below 10 ug/L. The decreasing trend in Mo continues between the Crows Landing and Vernalis sites, due to dilution from Merced, Tuolumne and Stanislaus River inflows. Only two samples collected downstream of the Merced River have ever exceeded 10 ug/L and both were collected at Crows Landing, one sample on January 3, 1986 (14 ug/L) and the other on May 25, 1997 (15 ug/L).

Conclusions

A primary concern for Mo is its ability to bioaccumulate in certain plant species, notably legumes, which in turn can cause molybdenosis and infertility in cattle. Symptoms of molybdenosis have been known to occur in ruminants (particularly cattle) grazing on forage containing concentrations of Mo above 10 ppm. Higher levels of Mo in forage (20 - 100 ppm) may be tolerated depending on the Cu to Mo ratio. For cattle, a Cu:Mo ratio >2 seems to prevent symptoms of molybdenosis. The levels of Mo for the SJR could cause concerns for cattle use were it not for several other factors which reduce the potential hazard these levels pose to fertility. There is a significant potential problem in Tulare Lake and Buena Vista Lake Basins with the possible infertility of young cattle grazing on forage irrigated with saline/sodic waters that contain Mo.

Concentrations in subsurface drainage collection sumps in the DPA averaged 50 ug/L with many concentrations exceeding 100 ug/L (Chilcott et al., 1988). Waters that enter SGWD, Camp 13 Slough, and Agatha Canal have lower Mo levels with maximum concentrations not exceeding 34 ug/L, and mean and median concentrations remaining near or below the 10 ug/L water quality objective for forage and pasture irrigation. The objective appears to have been violated only when no subsurface drainage water or wetlands releases were entering Mud Slough (north), indicating that normal seepage or background flow in the Slough contains highly elevated levels of Mo (Westcot et al., 1992).

Mo concentrations are consistently higher at SJR sites upstream of discharges from the Grassland Watershed where the flow consists largely of groundwater accretion. Only at the Lander Avenue site have Mo concentrations exceeded the 19 ug/L mean-monthly objective and the 50 ug/L maximum objective, background concentrations with no other source except groundwater accretion.

Even though Mo is not a human carcinogen, a guideline of 35 ug/L exists for potential human health concerns. For animals, a guideline of 10 ug/L exists for forage irrigation, whereas for drinking water it is 50 ug/L. As no effect from Mo bioconcentration in aquatic organisms has been observed.

Table III-2 Molybdenum Concentrations in Selected Water Bodies Grassland within the Watershed and San Joaquin River

	Time Period	Mo Concentration (ug/L)				Ratio
Site	of Record	Min	Mean	Median	Max	Cu:Mo
Upstream Subsurface Agricultural Drain						
Panoche Drain	09/85 - 11/96	2	8	8	16	1.4
Firebaugh (Main) Drain	09/85 - 11/96	<5	26	19	122	0.9
Water Bodies Entering GWD						
Camp 13 Slough	09/85 - 01/95	1	11	9	34	2.0
Agatha Canal	09/85 - 01/95	1	8	7	27	2.3
Water Bodies Exiting GWD						
Mud Slough Upstream of SLD	01/94 - 11/96	6	8	7	13	0.82
Mud Slough @ Hwy. 140	12/85 - 09/90	4	13	11	187*	0.51
Mud Slough @ San Luis Drain	10/90 - 01/97	<1	19	18	50	0.74
Mud Slough @ Lander Ave.	12/85 - 01/97	2	8	7	29	0.74
San Luis Drain @ Terminus**						
Pre-Grassland Bypass	12/93 - 02/95	<1	92	86	419	0.08
Post-Grassland Bypass	10/96 - 01/97	22	29	29	35	---
San Joaquin River						
Lander Avenue	12/85 - 01/97	<1	20	15	74	1.2
Hills Ferry	12/85 - 01/97	<1	8	7	19	0.86
Crows Landing	12/85 - 01/97	<1	5	5	15	1.8
Airport Way (Vernalis)	12/85 - 01/97	<1	2	2	2	3.8

* occurred on 02/07/86; only value exceeding 32 ug/L during period of record.
** only 10 samples collected pre-bypass and 2 samples collected post-bypass; high value occurred on 11/22/94. Remaining values below 110 ug/L.
GWD = South Grassland Water District.
SLD = San Luis Drain.

IV Established Water Quality Objectives and Beneficial Uses for the San

Joaquin River

The 1990 Rainbow Report recommended A...controlled and limited discharge of drainage water from the San Joaquin Basin portion of the study area to the SJR, while meeting water-quality objectives@ (page 3). Therefore, meeting water-quality objectives is the guiding principle.

A definition and discussion of water-quality objectives can be found in AThe Water Quality Control Plan for the Sacramento River and San Joaquin River Basins@ (Basin Plan) by the California Regional Water Quality Control Board Central Valley Region, Third Edition - 1994 (CRWQCB, CVR, 1994). The Porter-Cologne Water Quality Control Act defines water quality objectives as A...the limits or levels of water quality constituents or characteristics which are established for the reasonable protection of beneficial uses of water or the prevention of nuisance within a specific area@ [Water Code Section 13050(k)].

Beneficial Uses of the Lower San Joaquin River System

The Basin Plan (CRWQCB, CVR, 1994) states that Abeneficial uses are critical to water quality management in California@ and identifies existing and potential beneficial uses of surface water. Beneficial uses as identified in the Basin Plan are listed in Table IV-1 Beneficial use categories and abbreviations as defined in the Basin Plan for the SJR or Delta include Municipal and Domestic Supply (MUN), Agricultural Supply (AGR) including Irrigation and Stock Watering, Industrial Service Supply (IND), Industrial Process Supply (PRO), Water Contact Recreation (REC-1), Non-contact Water Recreation (REC-2), Warm Freshwater Habitat (WARM), Cold Freshwater Habitat (COLD), Wildlife Habitat (WILD), Migration of Aquatic Organisms (MIGR), Navigation (NAV), and Spawning, Reproduction, and/or Early Development (SPWN).

Table IV-1

According to the Basin Plan, ASignificant points concerning the concept of beneficial uses are:

1. All water quality problems can be stated in terms of whether there is water of sufficient quantity or quality to protect or enhance beneficial uses.

2. Beneficial uses do not include all of the reasonable uses of water. For example, disposal of wastewater is not included as a beneficial use. This is not to say that disposal of wastewater is a prohibited use of waters of the State; it is merely a use which cannot be satisfied to the detriment of beneficial uses. Similarly, the use of water for the dilution of salts is not a beneficial use although it may, in some cases, be a reasonable and desirable use of water.

3. The protection and enhancement of beneficial uses require that certain quality and quantity objectives be met for surface and ground waters.

4. Fish, plants, and other wildlife, as well as humans, use water beneficially.@

Water Quality Objectives

The Basin Plan presented several important points that apply to water quality objectives. Among them are the following: (1) objectives may apply region-wide or be specific to individual water bodies or parts of water bodies; (2) site-specific objectives may be developed whenever the Regional Water Board believes they are important; (3) changes to the objectives can occur if experience demonstrates that the objectives are not appropriately set to protect beneficial uses, or new scientific information becomes available on the effects of water contaminants. Where the Regional Water Board determines it is infeasible for a discharger to comply immediately with such objectives, compliance shall be achieved in the shortest practicable period of time (determined by the Regional Water Board), not to exceed ten years after adoption of applicable objectives.

The Basin Plan for the SJR contains water quality objectives that apply to agricultural subsurface and other discharges. The focus in this report is on water quality objectives for salinity, selenium, boron, and molybdenum. These objectives are given in Table IV-2, specified by location and season of application.

The Regional Board amended the Basin Plan in 1996 for the control of agricultural subsurface drainage discharges containing selenium from the Grasslands Watershed in the Lower San Joaquin River Basin. A summary of selenium objectives is provided in Table IV-2, and expanded in Table IV-3 with the compliance time table for meeting selenium objectives.

Waste Discharge Requirements (WDR) for the Grasslands Bypass Channel Project were adopted in July 1998 by the California Regional Water Quality Control Board, Central Valley Region (see Appendix A). This project discharges most of the subsurface agricultural drainage generated in the Grassland Watershed to Mud Slough (North). The waste discharge requirements were issued to the San Luis and Delta-Mendota Water Authority and the U.S. Bureau of Reclamation and include discharge prohibitions, effluent limitations, discharge specifications, receiving water limitations and numerous provisions with a focus on selenium. The effluent limits include monthly and annual load limits that are the same as those presented in Table IV-2.

Table IV-2 SUMMARY OF WATER QUALITY OBJECTIVES FOR THE LOWER SAN JOAQUIN RIVER BASIN
(Water Quality Control Plan, San Joaquin River Basin, CVRWQCB, CVR, 1994; as amended in 1996)

	CONSTITUENT			
Location	Salinity (µmhos/cm)	Selenium[1,2] (µg/L)	Boron[2] (mg/L)	Molybdenum[2] (µg/L)
Vernalis	*MAXIMUM 30-DAY RUNNING AVERAGE* *1 April - 31 Aug.* **700 µmhos/cm** *1 Sept. - 31 March* **1,000 µmhos/cm**			
Mouth of Merced River to Vernalis		*MAXIMUM* **12 µg/L** 4-Day AVERAGE **5 µg/L**	*MAXIMUM* *15 March - 15 Sept.* **2.0 mg/L** *16 Sept. -14 March* **2.6 mg/L** *MONTHLY MEAN* *15 March - 15 Sept.* **0.8 mg/L** *16 Sept. -14 March* **1.0 mg/L** *Critical WY* **1.3 mg/L**	*MAXIMUM* **15 µg/L** *MONTHLY MEAN* **10 µg/L**
Sack Dam to Mouth of Merced River		*MAXIMUM* **20 µg/L** *4-Day AVERAGE* **5 µg/L**	*MAXIMUM* **5.8 mg/L** *MONTHLY MEAN* *15 March - 15 Sept.* **2.0 mg/L**	*MAXIMUM* **50 µg/L** *MONTHLY MEAN* **19 µg/L**
Salt Slough		*MAXIMUM* **20 µg/L** *MONTHLY MEAN* **2 µg/L**	*MAXIMUM* **5.8 mg/L** *MONTHLY MEAN* *15 March - 15 Sept.* **2.0 mg/L**	*MAXIMUM* **50 µg/L** *MONTHLY MEAN* **19 µg/L**
Mud Slough (north)		*MAXIMUM* **20 µg/L** *4-Day AVERAGE* **5 µg/L**	*MAXIMUM* **5.8 mg/L** *MONTHLY MEAN* *15 March - 15 Sept.* **2.0 mg/L**	*MAXIMUM* **50 µg/L** *MONTHLY MEAN* **19 µg/L**

1. Refer to Table IV-3 and text for more detail including compliance schedule.
2. Selenium, boron and molybdenum are total concentrations.

Table IV-3 SUMMARY OF SELENIUM WATER QUALITY OBJECTIVES AND COMPLIANCE TIME SCHEDULE (Water Quality Control Plan, San Joaquin River Basin, CVRWQCB, CVR, as amended in 1996)

Selenium Water Quality Objectives (in Bold) and Performance Goals (in italics)

	Applies No Later Than			
Water Body/Year Type	10 January 1997	1 October 2002	1 October 2005	1 October 2010
Salt Slough and Wetland Water Supply Channels	**2 µg/L monthly mean, 20 mg/L maximum**			
SJR below the Merced River to Vernalis; Above Normal and Wet Water Year Types	**12 µg/L maximum**	*5 µg/L monthly mean*	**5 µg/L 4-day avg.**	
SJR below the Merced River to Vernalis; Critical, Dry, and Below Normal Water Year types	**12 µg/L maximum**	*8 µg/L monthly mean*	*5 µg/L monthly mean*	**5 µg/L 4-day avg.**
SJR above the Merced River to Sack Dam and Mud Slough (north)	**20 µg/L maximum**			**5 µg/L 4-day avg.**

[1] The water year classification will be established using the best available estimate of the 60-20-20 San Joaquin Valley water year hydrologic classification (as defined in Footnote 17 for Table 3 in the SWRCB=s *Water Quality Control Plan for the San Francisco Bay/Sacramento-San Joaquin Delta Estuary*, May 1995) at the 75 percent accedence level using data from the Department of Water Resources Bulletin 120 Series. The previous water year=s classification will apply until an estimate is made of the current water year.

V. 1990 Management Plan Recommendations

In 1990, the San Joaquin Valley Drainage Program (SJVDP) recommended a plan for management of subsurface drainage and drainage related problems (SJVDP, 1990). The Management Plan (1990 MP) recommendations for the Grassland subarea included the following components:

$ Source Control on 93,600 acres of irrigated land.

$ Reuse of drainage water on 2,600 acres of salt tolerant trees and halophytes (mostly Water Quality Zone A).

$ Operation of 120 acres of evaporation ponds and 130 acres of solar ponds.

$ Pumping approximately 8,000 acre-feet from the semiconfined aquifer under about 10,000 acres of land (Water Quality Zones A and B).

$ Retiring 3,000 acres of irrigated lands (Water Quality Zone A).

$ Discharging about 102,000 acre-feet of drainage water to wetlands and/or the SJR (while meeting water quality standards).

The 1990 MP divided shallow groundwater areas within the subareas into water quality zones to assist in development of drainage management plans. The zones were defined by their unique shallow groundwater-quality characteristics. Within the Grassland subarea, Water Quality Zone A generally corresponds to the 90,000 acre agricultural area which has subsequently been referred to as the Drainage Project Area (DPA) (RWQCB, Feb., 1998). This area is particularly characterized by elevated selenium levels in its subsurface drainage discharges.

The 1990 MP estimated both a potential drainage volume and a drainage volume requiring discharge after implementation of management measures. Potential drainage volume is that volume requiring management which would be collected and removed if all the land needing drainage was installed with drainage systems. Drainage volume requiring discharge is the potential drainage volume minus the reduction in volume achieved through implementation of drainage reduction measures, including source control, reuse, groundwater management, and changes in land use (such as land retirement). For Grassland subarea Water Quality Zone A, the estimated potential drainage volume was 54,000 af in the year 2000 and 56,700 af in 2040; with an average concentration of 5900 TDS, 11 ppm boron, and 160 ppb selenium (SJVDP Technical Information Record, Sept. 1990). After implementation of drainage management measures, the estimated drainage volume requiring discharge was 11,500 (10,700 + 800) af in 2000 and 21,200 (21,000 + 200) af in 2040.

For Water Quality Zone A, the 1990 MP recommended the following measures:

Grassland Subarea, Water Quality Zone A		
Drainage Management Measure	Estimated Drainage Volume Managed by Year 2000 (AF)	Estimated Drainage Volume Managed by Year 2040 (AF)
Source Control	24,000	25,100
Land Retirement	0	2,300
Groundwater Mgmt.	2,000	4,000
Drainage Reuse	16,500	4,100
Total	42,500	35,500
Evaporation System	800	200
Drainage Requiring Discharge to SJR	10,700	21,000
Total Potential Drainage Volume	54,000	56,700

Technical advances and progress in implementing drainage management measures are addressed in SJVDIP Subarea reports and other Technical Committee reports. This report addresses only drainage requiring discharge to the SJR.

For the drainage requiring discharge to the SJR, the 1990 MP included proposed facilities to isolate high selenium subsurface drainage from both irrigation spills and tailwater as well as regional wetlands, and to discharge the drainage through a reopened segment of the San Luis Drain (SLD). The new facilities would include (1) an extension of the SLD about 8 miles to the SJR below its confluence with the Merced River, (2) construction of a 7-mile bypass canal to convey drainage from Water-Quality Zone A to the SLD near Dos Palos, and (3) a cleanup of the SLD from Dos Palos to the present terminus. The estimated capital cost for the SLD extension was $16 million, based on an indexed USBR 1983 cost estimate for the SLD north of Kesterson. The estimated cost for cleaning the existing SLD was $6.8 million.

In determining the amount of drainage water that could be discharged, the 1990 MP focused on the assimilative capacity of the SJR. For planning purposes, the assumption was made that if the selenium water quality objectives were met, then generally the boron and salt objectives would also be met. The analysis assumed a 5 ppb (monthly mean) Se objective for the SJR. Water year 1986-87 monthly flow and water quality data for the SJR and for Mud and Salt Sloughs were used, rather than an annual mean over a time series. The 1986-1987 water year hydrology was considered typical of a dry year, providing less river assimilative capacity than either normal or wet years. The increase in amount of drainage that could be discharged (10,700 af in 2000 to 21,000 af in 2040) was assumed to result from reductions in salt and trace element concentrations in the drainage after a 50 year leaching period. The 1990 MP did not address the control or timing of the discharges.

VI. Developments Since the 1990 Management Plan

VI.A. San Luis Unit Drainage Program

In 1991, the USBR San Luis Unit Drainage Program proposed a new plan, based on the model results, to further develop the 1990 MP. The plan=s operational strategy involved controlling drainage discharges to the SJR according to the River=s capacity to assimilate regulated constituents, such as salt, selenium, and boron, which posed periodic constraints to existing patterns of discharge. The USBR model assumed perfect forecast and response to receiving water assimilative capacity and that the water quality of irrigation water and groundwater pumpage remained constant over the simulation period. The plan included the Grassland Bypass and San Luis Drain extension to the SJR similar to the 1990 MP. The proposed means of discharge control was implementation of a combination of discharge and source control measures, principally blending drainage with fresh water and then recycling for irrigation. The amount of recycling would vary according to River conditions. Under a worst case scenario, such as during drought periods of low River flows, most drainage would be recycled, in effect storing salts in the root zone. Conversely, when conditions in the River were favorable, recycling would be suspended and accumulated salts would be leached and discharged. Proposed recycling systems generally consisted of pumping facilities located at points where drainage could be conveniently collected, storage reservoirs to regulate drainage discharge, and pipelines to convey drainage to locations for blending with irrigation water and input into the distribution system. Total capital costs, including irrigation system improvements, drainage recycling facilities, Bypass and Drain extension, and land acquisition for alternative land use was estimated at $58.4 million.

District recycling operations were simulated using the Irrigation and Drainage Operations (IRDROP) model. Specified cropping patterns, shallow groundwater conditions, drained acreage, irrigation practices, and other parameters were input into the model and irrigation water demands, drainage production, discharge, recycling, and other operations were simulated. The analysis used data from a 30-year hydrologic period (1961-1990), and operated on a monthly time step.

The San Luis Unit Drainage Program applied only to federal contract districts, representing approximately 83 percent of the total drained area in the DPA. Consequently, the assumption was made that only 83 percent of the River=s assimilative capacity would be utilized. After implementation of the source control and alternative land use features of the plan, the estimated average total drainage discharged on an annual basis was projected to be 14,500 af. Regulating reservoir capacity was to be 2,050 af. The model analysis indicated that recycling frequency (the percentage of months over the 30-year analysis period in which recycling would be required) would vary by district from 4 percent to 32 percent of all months. The analysis further assumed that approximately 14,650 af of CVP water, conserved through source control and alternative land use on 8,000 acres, would be discharged to the SJR via the Newman Wasteway to augment flows for increased assimilative capacity.

VI.A2. CVRWQCB Model

Another screening level model developed by the CVRWQCB (Karkoski, 1995 unpublished analysis) considered load restrictions and model and response errors on the sizing of regulating reservoirs. Model and response errors were expressed by allowing only 80 percent of the available assimilative capacity to be used. When evaporation effects were considered, the storage size required for regulating reservoirs was found to be 26.8 million cubic meters. The large difference in regulating reservoir volume (4.3 vs. 26.8 million cubic meters) is a function of the different assumptions made in the two modeling approaches. Whereas in the USBR model, the full assimilative capacity of the river was available and there was no annual selenium load cap imposed, the CVRWQCB model assumed suboptimal use of the assimilative capacity and imposed the CVRWQCB Basin Plan=s annual selenium discharge load cap of 3,624 kg (CVRWQCB, 1996). The CVRWQCB model also assumed that a mean annual discharge of selenium from the agricultural water districts to the SJR was 2,945 kg. Although the above models differed in certain assumptions, the premise shared by both models was that regulating reservoirs could be constructed and managed to respond to real-time conditions in the SJR. In contrast, the analysis used by the CVRWCB in developing its control plan for selenium was based on a modified EPA load setting methodology (Karkoski, et.al., 1995; CVRWQCB, 1994) which assumes extremely limited ability to forecast, and therefore respond to, available assimilative capacity. The monthly flow record (1970-1991) was divided into eight flow regimes which differed based on water year type (dry and wet) and season. The selenium effluent limits were set for the low flow conditions in each flow regime (quasi-steady state) to meet an Aallowable@ rate of violation-once every three years as allowed by federal regulation.

Table VI-1 compares the annual allowable selenium load from the CVRWQCB analysis for dry years and wet years using dynamic (real-time) versus quasi-steady state modeling assumptions. As can be seen in this table the advantages to the discharger of using a real-time system are significant with respect to the amount of selenium load that could be discharged.

Table VI-1. Comparison of real-time and quasi-static selenium load limits

	WET YEAR Se LOAD (kg)	DRY YEAR Se LOAD (kg)
QUASI-STATIC	1,405	455
DYNAMIC (REAL-TIME)	3,364	2,105

VI.B. Grassland Bypass Project and Water Quality Monitoring, 1996-1997

Grassland Bypass Project

A Use Agreement (UA) was signed on November 3, 1995 between USBR and the San Luis Delta-Mendota Water Authority. The UA allows use of a 28-mile segment of the SLD to convey agricultural drainage from approximately 38,700 acres of tile-drained agricultural land to the SJR via a six-mile segment of Mud Slough (north). In September 1996, agricultural drainage from the DPA was diverted from Panoche and Firebaugh (Main) Drains to Mud Slough (North) via the Grassland Bypass. Prior to the use of the Grassland Bypass, agricultural drainage water was transported via Camp 13 Slough, Agatha Canal and Grassland Water District canals into either Salt Slough or Mud Slough (North). The UA allows for the initial use of the SLD for a two-year duration (Water Years 1997 and 1998) and allows for renewal of this interim use for no more than three years if certain conditions are met. For the present, the implementation of the Grassland Bypass Use Agreement has effectively removed agricultural drainage from Salt Slough and wetland water delivery channels in the Grassland area.

During the first year of the Grasslands Bypass project, considerable investment has been made by water districts in the Grasslands Basin in facilities to allow recycling of subsurface drainage water and to prevent commingling of tailwater and subsurface drainage. Sumps have been retro-fitted with controllers to allow drainage systems to be shut down during high rainfall-runoff periods, allowing more control over drainage discharge and mass loading of salts and other contaminants. Continued investment in these sort of technologies and reactive management to continually refine the operation of these systems will be needed to achieve SJVDP goals.

Among the renewal conditions are specific selenium load limits (Table VI-2) and development of a long-term drainage management plan for implementation consistent with the CVRWQCB Basin Plan Amendment. Effective January 10, 1997, the discharge of selenium from agricultural subsurface drainage systems in the Grassland Watershed was prohibited in amounts exceeding 8,000 lbs/year. The 1996 Basin Plan Amendment also states that effluent limits established in waste discharge requirements were to be applied to the discharge of subsurface drainage water from the Grassland Watershed.

Waste Discharge Requirements (WDR) for the Grasslands Bypass Channel Project were adopted in July 1998 by the California Regional Water Quality Control Board, Central Valley Region (see Appendix A- *available only in hard copy*). This project discharges most of the subsurface agricultural drainage generated in the Grassland Watershed to Mud Slough (North). The waste discharge requirements were issued to the San Luis & Delta-Mendota Water Authority and the U.S. Bureau of Reclamation and include discharge prohibitions, effluent limitations, discharge specifications, receiving water limitations and numerous provisions with a focus on selenium. The effluent limits include monthly and annual load limits that are the same as those presented in Table VI-2.

TABLE VI-2 SELENIUM LOAD VALUES (LBS)

	Year 1-2	Year 3	Year 4	Year 5
OCT	348	348	348	348
NOV	348	348	348	348
DEC	389	389	389	389
JAN	533	506	479	453
FEB	866	823	779	736
MARCH	1,066	1,013	959	906
APRIL	799	759	719	679
MAY	666	633	599	566
JUNE	599	569	539	509
JULY	599	569	539	509
AUG	533	506	480	453
SEPT	350	350	350	350
12-MONTH TOTAL[1]	7,096	6,813	6,528	6,246
ANNUAL LOAD TARGETS	6,660[2]	6,327[3]	5,994[4]	5,661[5]

1. The 12-month total for any given year is somewhat higher than the annual load target for that year because the monthly targets for the months of September, October, November and December have been adjusted to allow for greater selenium discharge than would typically occur. This adjustment has been made to provide greater selenium management flexibility during months when the assimilative capacity of the river is sufficient to sustain this greater load.

2. The annual 2nd year load target is based on the average annual loads discharged over a 9-year historical period (1986-1994) which includes both wet and dry year data, as well as full and partial water supply data. It is divided by month based on the average historical distribution of selenium loads except where the Total Maximum Monthly Load (TMML) calculation (using a 1-in-5 month violation rate) allows for a greater monthly load.

3. The 3rd year annual load target is based on a 5 percent reduction of the average historical loads. The 5 percent reduction is applied equally across all months except where the TMML (using a 1-in-5 month violation rate) allows for greater monthly selenium loads.

4. The 4th year annual load target is based on a 10 percent reduction of the average historical loads. The 10 percent is applied equally across all months except where the TMML (using a 1-in-5 month violation rate) allows for greater monthly selenium loads.

5. The 5th year annual load target is based on a 15 percent reduction from the average historical load. The 15 percent is applied equally across all months, except where the TMML (using a 1-in-5 month violation rate) allows for greater monthly selenium loads.

Grassland Bypass Project Water Quality Monitoring, 1996-1997

CVRWQCB monitoring sites for the GBP are on Mud Slough (North), Salt Slough and four SJR locations schematically shown in Figure VI-1. The Lander Avenue monitoring site is upstream of the confluences of Mud Slough (North) and Salt Slough. The Hills Ferry site is upstream of the Merced River, but downstream of the Sloughs. The Crows Landing site is downstream Merced River. The Airport Way near Vernalis site is downstream of the three major east-side tributaries.

Concentrations

Water quality changes resulting from GBP implementation can be observed (Figure VI-2, by looking at the changes in drainage constituents, electrical conductivity (EC), boron (B) and selenium (Se), in Mud Slough (North) and Salt Slough for the period prior to the GBP (Water Year 1996) and the first year following GBP operation (Water Year 1997). Both EC and B concentration declined in Salt Slough and increased in Mud Slough (North) after the GBP began operation. However, the most dramatic change occurred with Se concentrations. Removing agricultural subsurface drainage from Salt Slough reduced the selenium concentration to below 2.0 _g/L during Water Year 1997 as opposed to a range of 1.0 to 33.5 _g/L during Water Year 1996. A corresponding increase was seen in Mud Slough (North) with selenium concentrations ranging from 5.0 to 79.6 _g/L during Water Year 1997. The higher overall Se and B concentrations observed in Mud Slough (North) as compared to Salt Slough when subsurface agricultural drainage is present is due to the limited dilution potential in Mud Slough (North) which has a lower baseline flow (Chilcott, et al., 1998b).

Figure VI-3 compares EC, B and Se concentrations at three monitoring sites on the SJR: upstream of the Merced River (Hills Ferry), immediately downstream of the Merced River (Crows Landing) and downstream of the east-side tributaries (Airport Way near Vernalis) for Water Years 1996 and 1997 (Chilcott, et al., 1998a). The three sites follow similar trends, with concentrations decreasing downstream of the Merced River inflow at the Crows Landing site, and further reductions in concentrations occurring downstream of inflows from the Tuolumne and Stanislaus Rivers, at the Airport Way site near Vernalis. EC, B and Se concentrations are lowest when inflows from major east-side tributaries (Merced, Tuolumne and Stanislaus Rivers) are highest and provide the most dilution. Other than a sharp increase during March 1997 at the Hills Ferry site, concentrations peak during the irrigation season (April through August), a period of dry weather and little upstream dilution flow.

FIGURE VI-1

FIGURE VI-2

FIGURE VI-3

Loads

Monthly loads of salt for the SJR at Crows Landing and Vernalis (at Airport Way) are presented in Figure VI-4 for Water Years 1996 and 1997. Monthly salt loads in Water Year 1997 were generally the same or slightly lower than Water Year 1996 loads at both sites except during the flood months of January and February. Salt loads in January and February 1997 appear to have been much higher than the previous year, but flood flows and limited data availability during the 1997 flood makes flow and load estimates suspect. March 1997 salt loads are somewhat higher than March 1996 loads at Vernalis. (Chilcott, et al., 1998a).

Figures VI-5 and VI-6 show the monthly loads of boron and selenium, respectively, for the SJR at Crows Landing and Vernalis (at Airport Way) for Water Years 1996 and 1997. Boron loading at the two sites has a similar pattern to salt loading but the pattern of selenium loading is somewhat different. Selenium loads at both sites were 40 to 70 percent lower during the June to September period of Water Year 1997 than the same months in Water Year 1996. Loads were slightly lower at Crows Landing and up to 30 percent lower at Vernalis during the October through December period of Water Year 1997 as compared to Water Year 1996. Although flood flows and limited data availability during the 1997 flood makes selenium load estimates for January and February suspect, the limited data suggests that selenium loads during January 1997 at Crows Landing and Vernalis were more than double the prior year=s load. Selenium loads were also higher in February 1997 than in February 1996 at the Vernalis site.

FIGURE VI-4

FIGURE VI-5

FIGURE VI-6

VI.C. Grassland Bypass Project Biologic Effects Monitoring, 1996-1997

The Grassland Bypass Project Monitoring Plan (USBR et al., 1996) sets the locations and lays out the process for data collection to determine achievement of GBP objectives and compliance with agreed terms and conditions. GBP benefits include removal of agricultural drainage from Grassland Water District wetland delivery channels (including Salt Slough) to allow for full wildlife refuge management water deliveries, the reduction of fish and wildlife exposure to Se in wetland delivery channels, combining drainage flows into one structure to provide for more effective drainage management through improved measurement of flow and Se load, and establishment of an accountable drainage entity. These benefits are expected to outweigh the risks of degrading six miles of Mud Slough with drainage discharges and historically unprecedented levels of Se and other constituents.

Interpretations of Se effects on biota within the GBP monitoring area were based upon established ecological risk guidelines (Engberg et al., 1998) available for biologically relevant matrices collected for this project (Table VI-3). The three risk levels identified are the no effect level, the level of concern with unknown effects, and the toxicity threshold which represents the level at which reproductive impairment has been reported for warmwater fish species.

The initial results of biological monitoring since the re-opening of the drain have generally met expectations. The lower portion of Mud Slough (North), below the drainage discharge point, has suffered a measure of degradation. Certain fish species (mosquitofish, inland silverside, fathead minnow) from Mud Slough (North) at first accumulated Se to levels well above the toxicity threshold; more recently, these concentrations have declined somewhat, but are still above pre-project levels. Other species have increased Se body burdens (concentrations) yet remain within the level of concern zone. Crayfish were the only invertebrate within Mud Slough (North) to increase Se body burden levels to above the no effect level after GBP implementation. An ecosystem hazard assessment (Beckon et al., 1998) indicates that the Se hazard in Mud Slough (North) has risen marginally from low -moderate to high-moderate since initiation of the GBP.

Salt Slough, a wetland water supply channel from which subsurface drainage was removed as a result of GBP implementation, is showing signs of improvement with respect to Se levels in the biota. After one year of GBP operation, the 7 percent of all sampled fish species that had been within the toxicity threshold zone, fell to within the level of concern or no effect zones. Similarly, the 77 percent of all sampled invertebrate species that had been within the level of concern zone fell to within the no effect zone as a result of the GBP. An ecosystem hazard assessment (Beckon et al., 1998) indicates that the Se hazard to the aquatic ecosystem in Salt Slough has declined from high to low since initiation of the GBP.

As expected, the San Joaquin River reach between the Salt Slough and Mud Slough North confluences appears to be realizing project benefits with respect to Se levels in biota. After one and a half years of operation, consistent with site-specific water quality improvements, available data reveals Se levels in all fish and invertebrate species have declined and are remaining within the no effect zone associated with no ecological risk.

The SJR at Hills Ferry, just downstream of the drainage discharge point, was not expected to realize any GBP risks or benefits. The response of invertebrate species to GBP changes reflects this expectation, with post-GBP Se levels remaining consistent with pre-GBP levels, and with all invertebrates remaining within the no effect zone. Response of fish species is slightly different. After one year of project operation, fish samples within the no effect zone increased from 57 percent pre-GBP to 83.3 percent post-GBP , coinciding with a decline of fish samples in the level of concern zone from 43 percent pre-GBP to 16.6 percent post-GBP. This apparent improvement with respect to Se levels in biota in the SJR should be interpreted cautiously since certain factors that affect SJR water quality (i.e. dry or critical water year types) can influence biotic response to Se bioaccumulation, and have yet to be realized since the GBP was initiated. It is important to note that post-GBP biological data presented to date represents 1 2 years of a 5 year monitoring program; the additional 3 2 years of continued monitoring will provide the trend information necessary to identify the overall environmental risk potential of the GBP to the Mud Slough (North) and SJR systems.

Table VI-3.

Grassland Bypass Project Biomonitoring Results Summary: Percentages of Fish and Invertebrate Samples within Ecological Risk Categories Based on Selenium Concentrations in Tissue. (Data *in* Beckon et al., 1998; CDFG, 1998 *in draft*).

FISH[1] (all species)	***Ecological Risk Guidelines (Fish; whole body)***					
	No Effect *<4 Se mg/kg (dry wt)*		***Level of Concern*** *4-12 Se mg/kg (dry wt)*		***Toxicity Threshold*** *>12 Se mg/kg (dry wt)*	
	Pre-project[2]	Post-project[3]	Pre-project	Post-project	Pre-project	Post-project
Mud Slough (North)[4]	67.5	8.3	27.5	58.4	5.0	33.3
Salt Slough	11.6	40	81.4	60	7.0	0.0
San Joaquin River at Fremont Ford[5]	43	100	57	0.0	0.0	0.0
San Joaquin River at Hills Ferry[6]	57	83.3	43	16.7	0.0	0.0

INVERTEBRATES[7] (all species)	***Ecological Risk Guidelines (Invertebrates; whole body; as food chain item)***					
	No Effect *<3 Se mg/kg (dry wt)*		***Level of Concern*** *3-7 Se mg/kg (dry wt)*		***Toxicity Threshold*** *>7 Se mg/kg (dry wt)*	
	Pre-project	Post-project	Pre-project	Post-project	Pre-project	Post-project
Mud Slough (North)	74.1	57	14.8	43	11.1	0.0
Salt Slough	23	100	77	0.0	0.0	0.0
San Joaquin River at Fremont Ford	50	100	50	0.0	0.0	0.0
San Joaquin River at Hills Ferry	100	100	0.0	0.0	0.0	0.0

1. Fish species: mosquitofish, inland silverside, fathead minnow, carp, white catfish, channel catfish, bluegill, red shiner, threadfin shad, green sunfish, largemouth bass, Sacramento blackfish, striped bass, Sacramento squawfish, bigscale logperch; all samples composite whole-body
2. Pre-project samples were collected prior to September 1996 (March 1992 to August 1996).
3. Post-project samples represented were collected after September 1996; (November 1996 to March 1998); project sampling is continuing.
4. Mud Slough (North) data is combined for the two Project Sites downstream of the drainage discharge point.
5. The San Joaquin River at Fremont Ford is upstream of the Mud Slough (North) confluence, downstream of the Salt Slough confluence
6. The San Joaquin River at Hills Ferry is downstream of the Mud Slough (North) confluence, upstream of the

Merced River confluence.
7. Invertebrate species: red crayfish, water boatman, backswimmer, dragonfly, damselfly, giant waterbug; all samples composite whole-body.

VI.D. Real-Time Management of Drainage Discharge

The established water quality objectives presented in section IV are of numerical form, i.e. specify a concentration. Concentration represents the amount of mass per unit of water volume. Therefore, the mass (or load) of a constituent that can be discharged into a river without exceeding the water quality objectives is directly proportional to the flow of water in that river. The assimilative capacity of a water body is defined as the mass of a constituent that a receiving water can accept, without violation of the concentration limit at the compliance point for that constituent, at a given rate of discharge of both source and receiving water bodies. Water flow in the SJR varies with location along the River, time of year, and annual precipitation. Consequently, the assimilative capacity of the SJR is also variable.

The real-time water quality management concept is that the total load of a constituent discharged to the SJR that can be accommodated without impairing the beneficial uses of the water, is based on the ability to vary the time and place of discharge to match the temporal and spatial variability of the assimilative capacity. In the approach to real time water quality management, it is proposed to actively manage the assimilative capacity of the River by controlling discharge of salts from agricultural lands and wetlands. The present timing of discharges of dissolved solids and trace elements from the DPA and the timing of reservoir releases is such that the water quality objectives of the SJR are often exceeded at the compliance monitoring locations. Opportunities for adjusting the timing of discharges and reservoir releases have been identified (Figure II-2), although the practical constraints to making such adjustments have not been thoroughly explored. By making such adjustments, temporal variations in water quality could be minimized and the frequency of violation of water quality objectives could be reduced. A real-time water quality management system, along with constituent load reduction, could allow continued discharge of salt from agricultural lands and wetlands while minimizing the impacts on the SJR and minimizing violations of water quality objectives.

The goal of real-time management is to make multiple use of water that is already being stored or released for other purposes. For example, releases are currently being made from tributaries to the SJR for the explicit purpose of providing pulse/attraction flows for fish; releases are also being made from New Melones for the explicit purpose of providing dilution flows to meet water quality objectives at Vernallis.

Coordination of existing reservoir releases for fish flows with existing discharges of salt can have the net result of reducing reservoir releases needed explicitly to provide dilution flows. Real-time management as applied in this ex ample would result in less storage and fewer releases being made explicitly for providing dilution flows, although the total salt load to the SJR would not be reduced.

Present Activities

Opportunities for real-time management of drainage discharge are currently being explored. CALFED is funding a project by the San Joaquin River Management Program, Water Quality Subcommittee (SJRMP-WQS, comprised of members representing the DWR, CVRWQCB, and Lawrence Berkeley National Laboratory) to conduct studies of real-time management. Past analysis using mass balance models of the SJR suggest that considerable opportunity exists for improved coordination of drainage discharges and reservoir releases to more efficiently use SJR assimilative capacity for salts. The SJRMP-WQS was awarded a grant in 1994 to demonstrate that improved management and coordination of tributary releases and agricultural drainage from Grassland sources could significantly reduce the frequency of violations of water quality objectives for salinity, selenium, and boron in the SJR. The SJRMP-WQS developed a decision support system that retrieves current flow and water quality data and allows forecasts of SJR assimilative capacity to be made for salinity at Vernalis. These forecasts will become increasingly useful to water districts and other agencies for timing releases of discharge loads from agricultural lands, wetlands and wildlife refuges on the westside, in coordination with eastside reservoir flow releases for salmon migration, recreation, and water quality. SJRMP-WQS uses SJRIO model to forecast salinity concentration and load in the SJR.

Real-Time Data Acquisition Systems and Quantitative Evaluation

Real-time water quality management requires techniques that update the state of knowledge of a system continuously and allows actions to be taken to meet water quality objectives. Such techniques are being developed for the SJR Basin to promote voluntary compliance with state water quality objectives for priority contaminants such as salt, selenium, and boron. The SJRMP-WQS has established a real-time water quality monitoring network in the Grassland Basin with sites on the main stem of the SJR and tributaries to demonstrate the concept of real-time management of water quality. Ten sites were chosen for real-time monitoring of flow, EC, and temperature; they are listed in order from upstream to downstream, together with the sensor data collected at each site:

$ SJR at Lander Avenue (EC, flow, temp.)

$ Salt Slough at Highway 165 Bridge (EC, flow, temp.)

$ Grassland Bypass (compliance point - site B) (EC, flow, temp.)

$ Mud Slough near Gustine (EC, flow, temp.)

$ Merced River near Stevinson (EC, flow, temp.)

$ SJR at Newman (flow)

$ Orestimba Creek (EC, flow)

$ SJR at Crows Landing (EC, flow, temp.)

$ Patterson ID diversion (flow)

$ West Stanislaus ID diversion (flow)

$ SJR at Vernalis (EC, flow, temp.)

Although SJR stage, EC, and temperature are being monitored on a real-time basis, other real-time water quality monitoring is generally limited to those parameters such as dissolved oxygen for which no sample preparation is required. Techniques for the real-time measurement of other parameters of concern in the SJR, such as selenium and boron, have not been established nor are reliable sensors available.

The data from these stations is currently telemetered via modem to central data processing stations at the Lawrence Berkeley National Laboratory and DWR, where the information is checked for errors and missing values, and converted into a format accessible by a daily water quality forecasting model. The SJRMP-WQS developed a water quality forecasting model to provide 14-day forecasts of EC at Vernalis to assist water agencies such as the USBR-CVO in their daily operations to meet water quality objectives. The evolution of this model and its application is the nexus of water resources modeling activities for three agencies within California, and the University of California: USBR, DWR, CVRWQCB, and LBNL. The SJRMP-WQS also developed a website and public list server to improve communication between agencies and stakeholders concerned with SJR salinity.

The real-time water quality management system under development for the SJR Basin takes advantage of some of the features of the existing hydrologic data acquisition and forecasting programs. Unique aspects of the real-time water quality management system that are not replicated by current programs are:

1. Use of water quality sensors: currently only EC, temperature and pH are continuously logged, although there are a great number of constituents of concern within California=s river systems;

2. A continuous and integrated system of data error checking and validation because the data are used for regulatory purposes;

3. Addition of control systems that can be used to manage agricultural and wetland drainage water flow and water quality; and

4. Institutions that coordinate actions and responses of regulators, operators, and other public and private entities. Long-term commitment by agencies to support real-time data collection and water quality forecasting efforts.

San Joaquin River Daily Input-Output Model

In order to provide forecasts of flow and water quality in the San Joaquin River estimates are required of all the hydrologic inflows to and outflows from the San Joaquin River and the water quality associated with these inflows and outflows. These estimates are developed from historic data that were arranged as input data files to the monthly San Joaquin River Input Output Model (SJRIO-2).

The San Joaquin River Daily Input-Output model is a mass balance model which calculates daily flows and concentrations of total dissolved solids (TDS), boron, and selenium for a 96 km reach of the SJR from Lander Avenue to Vernalis (SWRCB 1985). An extensive database was assembled, with data for water years 1977 to 1985, to run the model. The model has been further modified to run on a daily time step so that it can be used with real-time flow and water quality data on the SJR. The daily model, SJRIODAY contains the following tributary river segments:

$ Six miles of Salt Slough below the Highway 165 gaging station.

$ Nine miles of Mud Slough (North) below the Gustine gaging station.

$ Five miles of the Merced River below the Stevinson gaging station.

$ Fifteen miles of the Tuolumne River below the Modesto gaging station.

$ Nine miles of the Stanislaus River below the Ripon gaging station.

$ Several miles of three west-side tributaries: Del Puerto, Orestimba and Hospital/Ingram Creeks.

These data are used to establish initial conditions for model runs and to generate a two-week forecast of flow and EC. Real-time data are supplemented by mean monthly flow and water quality data for other model components for which no real-time data are available, including: groundwater, riparian and appropriative diversions, surface and subsurface agricultural return flows, riparian evapotranspiration, evaporation, and precipitation. These components are estimated within the model based on seasonal variability and wet/dry water year classification provided by the modeler.

Estimate of Potential Impacts

Real time management of the SJR for salinity may involve drainage recycling (which may affect crop yields if root zone salinity is not carefully managed) and short-term surface storage, which could have negative impacts on wildlife if ponds are poorly designed or if water remains ponded during the waterfowl nesting season. This concept requires close cooperation between agencies that do not have a history of coordinated interaction -- hence some institution building will be required. Real-time management shifts the temporal distribution of salt loads. Therefore, there is a potential for increased concentration of salinity when load increases at a particular time which may result in an environmental impact.

Future Direction

CALFED has granted funding to the SJRMP-WQS real-time water quality management program for enhancement of the existing network of EC and flow monitoring stations and for improvement of model forecasting capability on the SJR and to increase confidence in flow and EC forecasts. The organizational infrastructure to manage agricultural subsurface drainage is currently being put into place. Outreach to potential cooperators to improve communication between drainage producers and eastside reservoir operators will continue. On-site infrastructure will need to be expanded to increase drainage storage capacity.

VII. Assessment and Recommendations

VII.A. Assessment

The concept of drainage discharge to the SJR as an option for drainage disposal from the Grassland Basin has undergone some refinement in the eight years since the publication of the recommendations of the SJVDP 1990 MP. The first improvement was documented in the San Luis Unit Drainage Program Plan, a more focused study of drainage options for westside agricultural areas receiving CVP water supplies. The analysis partitioned SJR assimilative capacity between CVP and non-CVP agricultural return flows and assumed that the approximately 83 percent CVP fraction of the assimilative capacity was fully available (Quinn et al. 1993). The analysis further recognized the need for short-term drainage holding ponds and facilities that would allow water-district scale subsurface drainage recycling if monthly drainage discharges from the Districts into the SJR would not meet State water quality objectives for selenium and boron.

In this report assimilative capacity means physical and chemical assimilative capacity for salts and trace elements. The assimilative capacity for a contaminant in the SJR is calculated as the difference between the current contaminant load in the River and the maximum allowable load, determined as the product of the design flow and the water quality objective for the contaminant. This discussion doesn=t include biological assimilative capacity which is unknown at the present time. In cases where the current contaminant load exceeds the allowable contaminant load there is no assimilative capacity. The calculation of assimilative capacity at a particular compliance point on the SJR or its tributaries assumes that the water quality objective is protective of all designated uses of the water at that point and all points downstream.

Quinn and Karkoski (1999) have recognized that the existing infrastructure is inadequate to control boron and selenium loads in drainage return flows in such a way that they would match SJR assimilative capacity. Needed Investments in technology and infrastructure would include:

$ a decision support system and monitoring network to produce accurate real-time forecasts of river assimilative capacity for salts and boron; and conducting scientific research on biotransformation of selenium to establish basis for real-time management of discharge to coincide with assimilative capacity of the system.

$ continuous-recording, development of telemetered sensors for selenium and boron;

$ real-time monitoring and control of drainage discharge at the water district level;

$ installed capability to separate and recycle tailwater and tilewater;

$ ability to store drainage in surface impoundments for short periods or in the ground by regulating tile drain flows;

$ centralized monitoring, coordination, and management of westside agricultural and wetland return flows and contaminant loads;

$ coordination and communication of operational decisions affecting eastside reservoir releases and SJR diversions.

In 1995, the San Joaquin River Management Program Water Quality Subcommittee began a two year project to demonstrate the potential for real-time water quality management on the SJR (Quinn et al, 1997). The Subcommittee developed a water quality forecasting model for salinity and graphic user interface for interpretation of model results and were able to show its capability for predicting SJR assimilative capacity for salt under a wide range of hydrologic conditions. The project was restricted to salinity management owing to the continuing lack of sensors to provide real-time data on water-borne selenium and boron concentrations.

Based on the information provided by members of the WQS, they were only partially successful in breaking down institutional barriers to allow better communication of flow and water quality information between decision makers on the SJR and along its major tributaries. The Subcommittee had difficulty persuading a single institution to Achampion@ the system that was developed and to utilize it to guide their own decision making. Other constraints to a real-time management scenario is lack of institutional structure and a water master for coordination of reservoir releases currently required for fish or water quality purposes and drainage discharges.

GBP was authorized in 1996 through a use agreement between USBR and representative of Grassland Area Farmers. The Use Agreement was developed based on input from State and federal agencies, agricultural, and environmental stakeholders. The Use agreement and the CVRWQCB Basin Plan prescribes a monthly and annual load limit for selenium discharges for the GBP. The load limit may not be appropriate for B and Mo, and may not be efficient for selenium either. A real-time management of discharge for salts, B and Mo and even selenium is possible, recognizing the progress that would need to be made before real-time water quality management could become a reality (Quinn, McGahan and Delamore, 1998). The current system of load-based targets is inefficient in its allocation of the river=s assimilative capacity and reduces operational flexibility.

In the two years since the GBP was initiated, considerable progress has been made towards the implementation of a real-time water quality management system. A comprehensive monitoring system has been developed with telemetry capability, allowing real-time access to flow and EC data at the inlet and outlet of the SLD, in Mud and Salt Sloughs, and at Crows Landing on the SJR. The Panoche and Firebaugh Drainage Districts have installed their own telemetered monitoring system at the outlet from the Districts, as well as within major conveyance facilities. Individual drainage sumps have outflow totalizing meters installed to measure tile drainage flows. Panoche Drainage District has installed a high capacity recirculation system that has the capacity to recycle all of the District=s tile drainage. Several Districts operate temporary holding ponds to allow some control on District discharge to the SJR. Most Districts within the Basin have adopted moratoria on tailwater discharges, allowing tilewater and tailwater to be separated. This reduces the total volume of drainage to be managed, improving control and operational flexibility. Recognizing the need for further progress, the CALFED Bay-Delta Program awarded a continuation grant to allow the Subcommittee to address some of the infrastructure and technology requirements identified above.

Although much of the effort in attempting to improve compliance with the GBP use agreement load limits for selenium in the San Joaquin River and its tributaries has focused on source control and adjustments to the timing of drainage releases and east-side reservoir releases the development of site specific or temporal selenium criteria should not be overlooked. These criteria would require considerable research and testing utilizing an adaptive approach but, in the long term, could benefit agriculture and the environment alike. The goal of this initiative would be to estimate a true assimilative capacity of the receiving water body that could then be expressed as a concentration criteria for the purposes of compliance monitoring. Hence seasonal criteria or criteria based on some other aspect of water biogeochemistry might conceivably be developed.

For salt, boron, and molybdenum, the conclusions drawn in Chapter 3 of this report, suggest that the concentration based management may be protective of most important beneficial uses. The argument made in drawing this conclusion is centered on the ability to readily define the users of water that are affected by waterborne concentrations above mean monthly and maximum threshold values. Hence the assimilative capacity of the SJR for each of conservative constituents (salts, B, and Mo) can be calculated with a reasonable degree of confidence, given reliable flow and concentration data. Selenium, on the other hand, has a more complex biogeochemistry than the other trace elements and has variable toxicity to invertebrates, fish, and higher order biota, depending on its chemical (redox) state. The redox state of selenium is readily transformed from the selenate form in oxidizing and high pH conditions to selenite, elemental selenium and selenide in reducing and low pH conditions. The toxicity of selenium to biota is also highly variable. Further complicating the toxicity thresholds among higher order biota, such as waterfowl, are the mechanisms of bioaccumulation and biomagnification of selenium, through which low doses of selenium in the diet can accumulate to toxic levels in tissue through continuous exposure.

Reported in Chapter 3 were the findings of an expert panel which concluded that waterborne selenium concentrations were not always a good measure of environmental toxicity. This conclusion would suggest site-specific and even seasonal water quality criteria. This method of development would allow a better determination of selenium species and the exposure pathways of selenium to at-risk biota would be an improvement over the current USEPA criteria for aquatic life that are fixed both in time and in space. Unfortunately, determination of selenium species is even more difficult and costly than determination of total waterborne selenium ; the technology for making these measurements is still in development. Simple models such as the function developed by Van DeVeer (1997) that combines measurements of total sediment selenium and organic carbon to obtain an index of selenium toxicity would be promising to advance the concept of site-specific objectives/criteria, if they can be shown to be easily implementable. At present, there has been insufficient testing of this model to suggest further development of this concept. Innovative approaches to develop reliable analogs for selenium toxicity are needed to advance the development of site-specific selenium objectives.

VII.B. Future Opportunities

To use the assimilative capacity of the River by real-time management, the monitoring network as well as the user base for the flow and water quality forecasts produced by the SJRMP-WQS should be expanded. Having participants actively engaged in the supply of data as well as being the recipients of the modeling results is fundamental to the success of the project. Techniques to enhance the accuracy of the forecasts should be explored and use of the Internet and other information delivery systems should be expanded. Institutions are reluctant to rely on a new service it proves unreliable or if it has a limited life.

For salts, B, and Mo assimilative capacity of SJR can be calculated and their discharged managed through a real-time monitoring and discharge.

The implications for the development of selenium objectives that are sensitive to temporal and spatial variability within the SJR system and its major tributaries has profound bearing on real-time water quality management of the SJR and the ability of westside drainage dischargers to meet selenium objectives. If site-specific and temporal selenium objectives could be raised without harm to the environment during those months when agricultural drainage discharges were highest, it could ease the burden on irrigated agriculture allowing more frequent attainment of objectives.

Conversely, if existing objectives were shown to be inadequate during certain periods that were critical for breeding or propagation of certain species of biota, more severe measures would need to be taken by agricultural dischargers during those critical periods to protect the environment. These measures might change from year to year.

Further progress needs to be made on source control, drainage reuse, drainage treatment technologies and other possible measures to reduce volume of drainage water and load of trace elements. Facilities for temporary storage of drainage water would also add flexibility to a real-time management strategy. These measures would reduce the amount of drainage water and trace elements to be discharged to the River.

There is not adequate assimilative capacity for salts, B, Mo, and selenium in the River upstream of Merced River confluence. The 1990 Plan recommended extension of the SLD to SJR below Merced River confluence because the River would provide greater assimilative capacity for discharge. In the context of real-time management, assimilative capacity is greater in the River if the SLD is extended as recommended in the 1990 Plan. However, consideration should be given to the following issues in any planning effort for such an extension:

$ Extension of the SLD to below Merced River confluence with the River could allow for increased opportunity for real-time management.

$ Potential changes in the USEPA selenium criteria could impact the decision process on the merit of the extension of the SLD.

$ Increase in subsurface drainage reuse or other methods of drainage reduction and drainage treatment could reduce the volume of drainage and the load of selenium needing disposal and thereby also affecting the relative merit of extension of the SLD.

$ A complete evaluation of the future projection of selenium load should be conducted.

$ Any planning for implementation of extension of the SLD should consider other options, cost/benefit analysis and an identification of tradeoffs. Data such as the Salt and Mud Slough biological data presented in Table VI-3 would facilitate the assessment.

VII.C. Study and Research Needs

$ Support selenium mass balance studies to measure sources and sinks of selenium and boron in aquatic ecosystems.

$ Research and field documentation of boron effects on crop yields should be encouraged. The objectives would be to quantify effects of salinity/boron interactions and transient boron levels, resulting from rainfall and/or sequential use of saline/sodic drainage waters for irrigation, on crop symptoms and crop yield.

$ Similarly, conduct research of sublethal and chronic impacts of boron and boron/salinity/toxic element interactions on fish and other aquatic species should be encouraged.

$ Since Mo levels of 5 mg/kg in forage can cause infertility in cattle, the committee recommends that researchers who work on use of saline/sodic drainage water to irrigate forage, particularly if the forage is to be grazed by young female cattle, should monitor the various aspects of this potential problem closely.

$ There is no information available on the effects of sulfates from the San Joaquin River on juvenile Chinook salmon during out migration and the smoltification process. It is also unknown how long these smolts remain in the San Joaquin River. Research is needed to determine the effects of salinity in the San Joaquin River on Chinook smolts and rearing time in the river.

$ Support and encourage research to develop continuous sensors for measuring selenium and boron concentrations in the San Joaquin River and its major tributaries. Sensors should be accurate within a reasonable range, robust enough for field installation and economical to produce.

$ Holding ponds could be designed that would prevent wildlife impacts but facilitate real-time management of drainage discharge.

$ Monitoring and research on long-term effects of selenium contaminated drainage water on lower SJR and the Delta is needed.

$ Ecotoxicity of Trace Elements

Among the trace elements of adverse impact in drainage discharged into the SJR, ecotoxicity of selenium compounds probably constitutes the most complex issue. The large gaps in knowledge has its roots in the extensive biogeochemical transformation and bioaccumulation of the selenium element, as already outlined above.

These research gaps were focused and publicized in the recent APeer Consultation Workshop on Selenium Aquatic Toxicity and Bioaccumulation@ held by the USEPA. The consensus opinion from the nine-member panel was that waterborne Se concentration is not always reliable indicator of Se adverse effects in aquatic top predators. This is because Se exposure and effects in top predators (the major concern for Se contamination) is mainly mediated through diets, i.e. the foodchain organisms in which biotransformations and bioaccumulation occur. The consensus opinion emphasizes the sediment and its resident food-chain organisms as a major sink for Se bioaccumulation and biotransformations. Since these biogeochemical processes are very complex, they may be highly variable from site to site, leading to the need to address Se impact on a site-by-site basis.

However, our present knowledge of these processes is inadequate to allow an extrapolation from waterborne Se concentrations to Se impact on top predators on a site-specific basis. Nevertheless, such extrapolation is needed for setting appropriate water quality criteria for different site conditions. For sustainable protection of water quality, research is also needed to assess the biogeochemical assimilatory capacity of a given system with respect to biological or ecological impacts; such impacts are the sole reasons of concern over trace elements such as Se.

The following recommendations are based on all of the above considerations, which emphasizes a better understanding of the sediment biogeochemistry and organoselenium pathway. The goal of this improved knowledge is the development of a more reliable yet practicable site-specific chronic water quality criteria for Se. The needed research topics are:

1. Biotransformations by primary Se fixation organisms that are omnipresent and unavoidable, such as microalgae and heterotrophic bacteria, both in the water column and sediment compartments.

    1) Research is urgently needed in these Se fixation processes that initiate the foodchain loading that leads to ecotoxic effects on top predators.

    2) An important key to remediation may be dissipation pathways at these lowest-trophic levels, such as volatilization. However, the value of the dissipation pathways to reducing ecotoxicity is currently controversial since Se fixation is integral to these pathways.

2. Se pathways from primary Se fixation through lower trophic consumers, to top predators such as fish.

    1) Determination and comparison of Se forms (e.g. free selenomethionine vs. proteineceous selenomethionine) or fractions in compartments (e.g. sediments, reproductive systems of top predators) of the highest bioaccumulation and ecotoxic risk, at each trophic level.

    2) Development of practicable analytical methods for the appropriate Se forms or fractions in these compartments.

3. Interactions of the above processes with different physical, chemical, and biological factors such as salinity, temperature, pH, light conditions, residence time, contaminant composition, and foodchain characteristics.

4. Determining and understanding the reasons for heterogeniety of sediment Se concentrations and forms.

1) Relationship to waterborne Se concentrations and to other sediment parameters such as total organic carbon.

2) Investigation on relationship of ecotoxic indicators (determined in 2) to waterborne and sediment Se concentrations.

3) These investigations should be aimed at facilitating correlation of past and existing water quality data with newly identified indicators.

5. Using the results from #1-4, identify and measure pertinent parameters that facilitate the development and validation of Se biogeochemical/ecotoxicity algorithms or models applicable to both lentic and lotic systems. Such model(s) should lead to a better prediction of site-dependent ecotoxic risk, and be able to feedback to real-time decision making and estimating biogeochemical assimilatory capacity with respect to ecotoxic risk.

6. Employing the algorithms(s) from #5, explore means to reduce Se ecotoxic risk in drainage waters prior to discharge into the SJR. As examples, investigate whether ecotoxic risk can be reduced by manipulating conditions in drainage storage reservoirs including invertebrate harvest to break Se foodchain transfer to top predators, enhancing Se volatilization while reducing incorporation of ecotoxic Se forms into biota, and optimization of reservoir configuration to reduce wildlife usage.

7. Support research to develop accurate techniques to measure in-situ selenium concentrations in both water and sediments. Develop analogs for selenium toxicity where possible to allow progress towards continuous monitoring of water-borne selenium concentrations.

8. Support research in improving techniques for in-situ and laboratory analysis of selenium speciation.

9. Conduct research to evaluate temporal and site-specific selenium criteria that is sensitive to variations in toxicity in time and/or in space.

VII.D. Recommended Actions

$ Support continued efforts of the SJRMP Water Quality Subcommittee to improve cooperation among diverters, dischargers and other beneficial users along the mainstem of the San Joaquin River and encourage coordination of their operations. Encourage SJRMP-WQS to find "champions" of the water quality forecasting system developed by the committee and to work with these organizations to tailor information to their operational needs. Encourage coordination among SJR users to facilitate the real-time management concept.

$ Continue support of essential water quality and flow monitoring stations along the San Joaquin River and its tributaries. Develop funding strategies to make the monitoring system self-supporting.

$ Continue developing drainage control and management strategies that will allow full participation in a continuous real-time management system in the SJR Basin. At the same time, develop the information needed to show that real-time management is at least as protective of the environment as the current load-based regulatory process. With this information, seek changes in the regulatory limits placed on selenium discharges.

$ Periodic evaluation of the real-time management of discharge to the River should be conducted to improve upon the concept.

$ Collection of the biological data obtained for the GBP (Table VI-3) should continue. Such data are critical in evaluating the need for and merit of the extension of the SLD as recommended in the 1990 Plan as an option for long-term drainage management in the Basin.

$ In planning process for extension of the SLD, other options and potential long-term economic and environmental benefits and impacts of such a project must be analyzed and evaluated.

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APPENDIX A

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Figure II-1 San Joaquin River Watershed from Mendota to Vernalis

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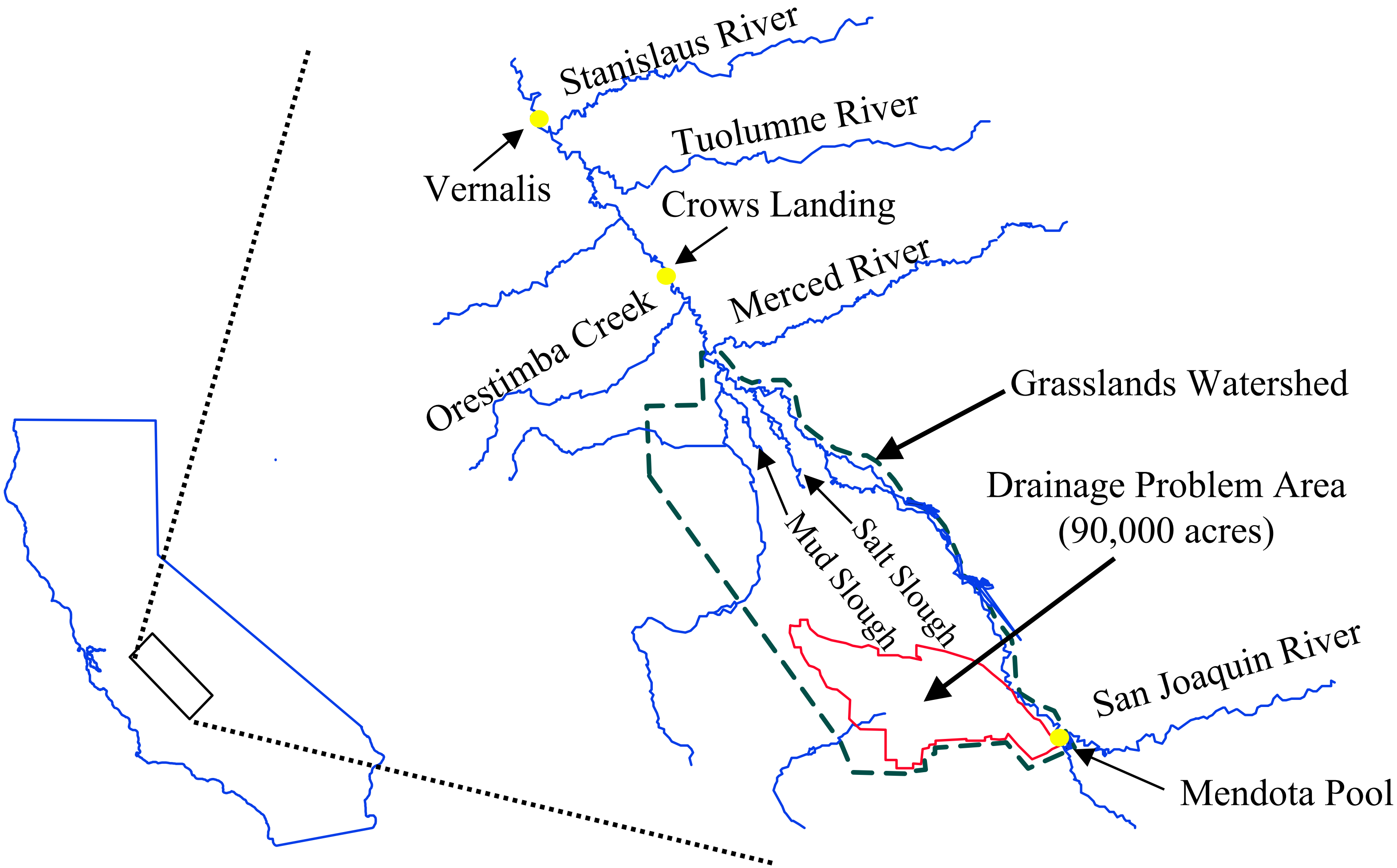


Figure II-2 San Joaquin River near Vernalis 30 Day Running Average Electrical Conductivity

(CRWQCB, CVR 1997)

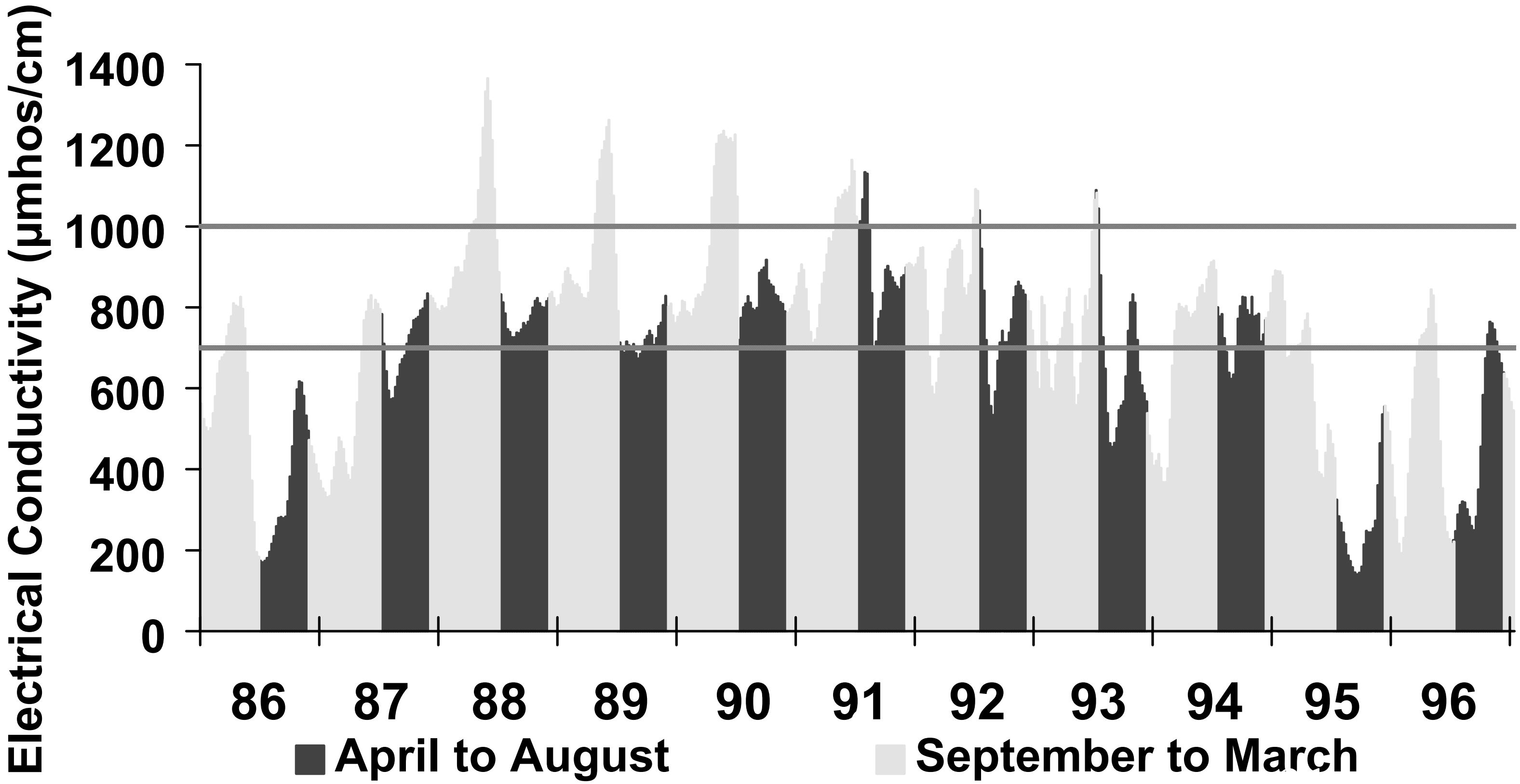
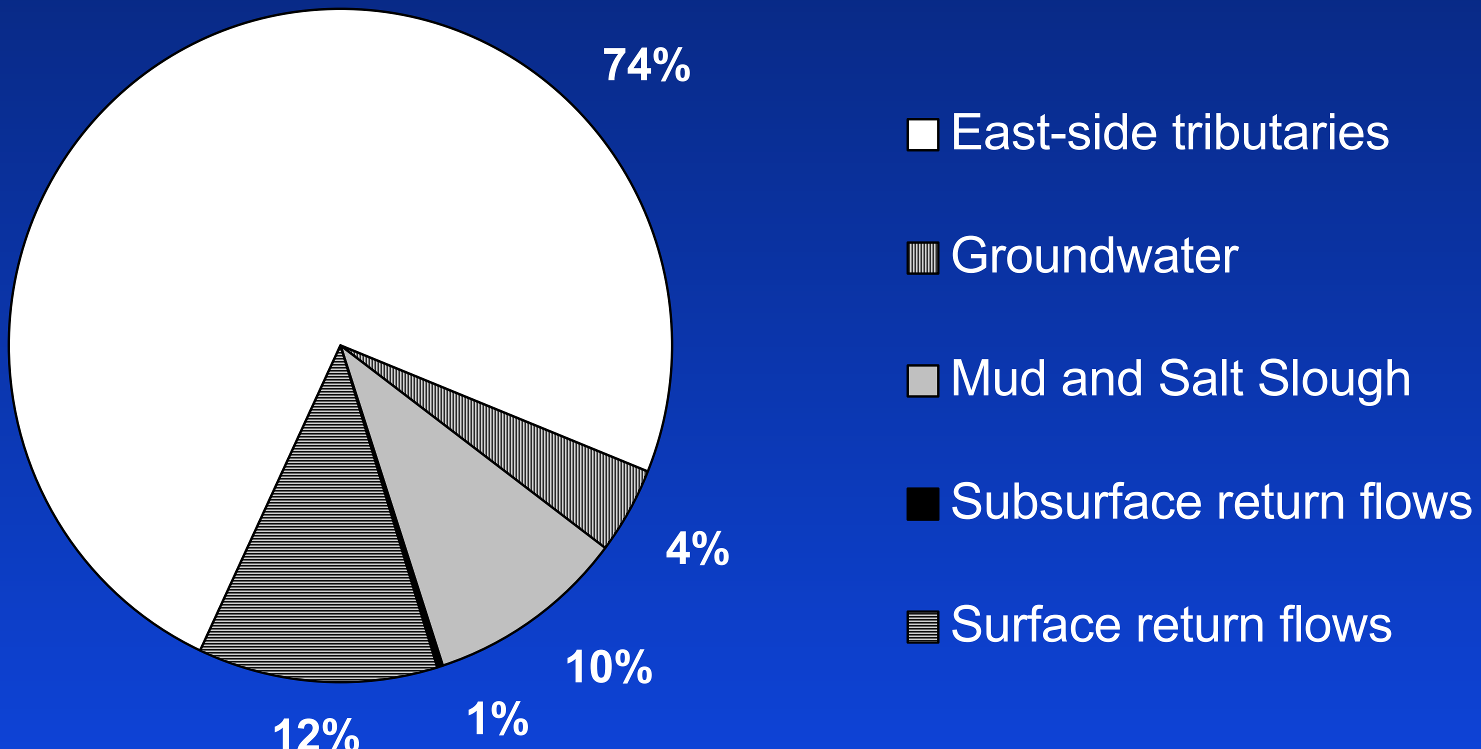


Figure II-3 Lower San Joaquin River Discharge



Mean Annual Discharge to SJR for WY 85 to 95: 2.2 million acrefeet

Based on combination of historical and SJRIO model data

Figure II-4 Lower San Joaquin River Discharge

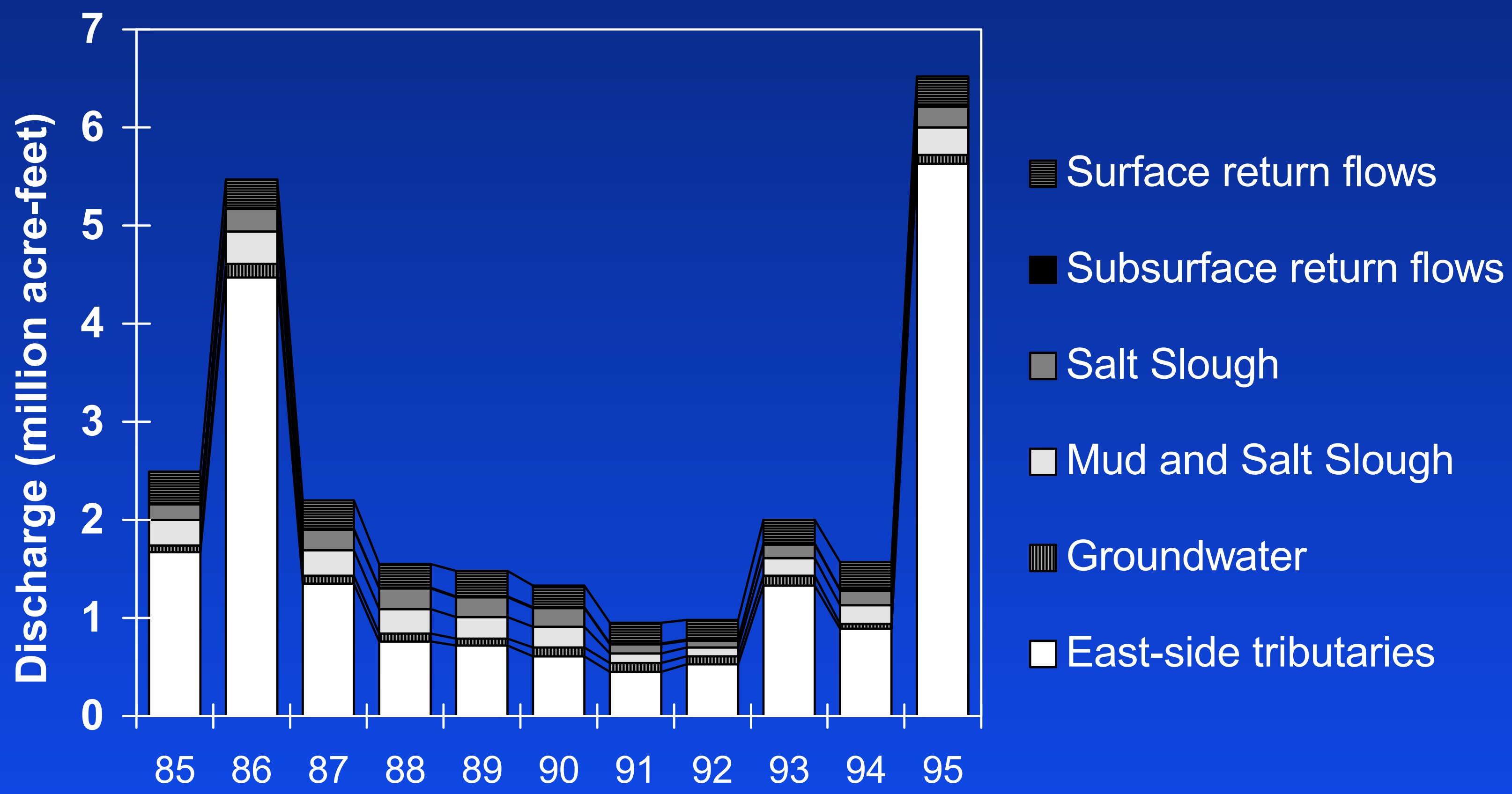
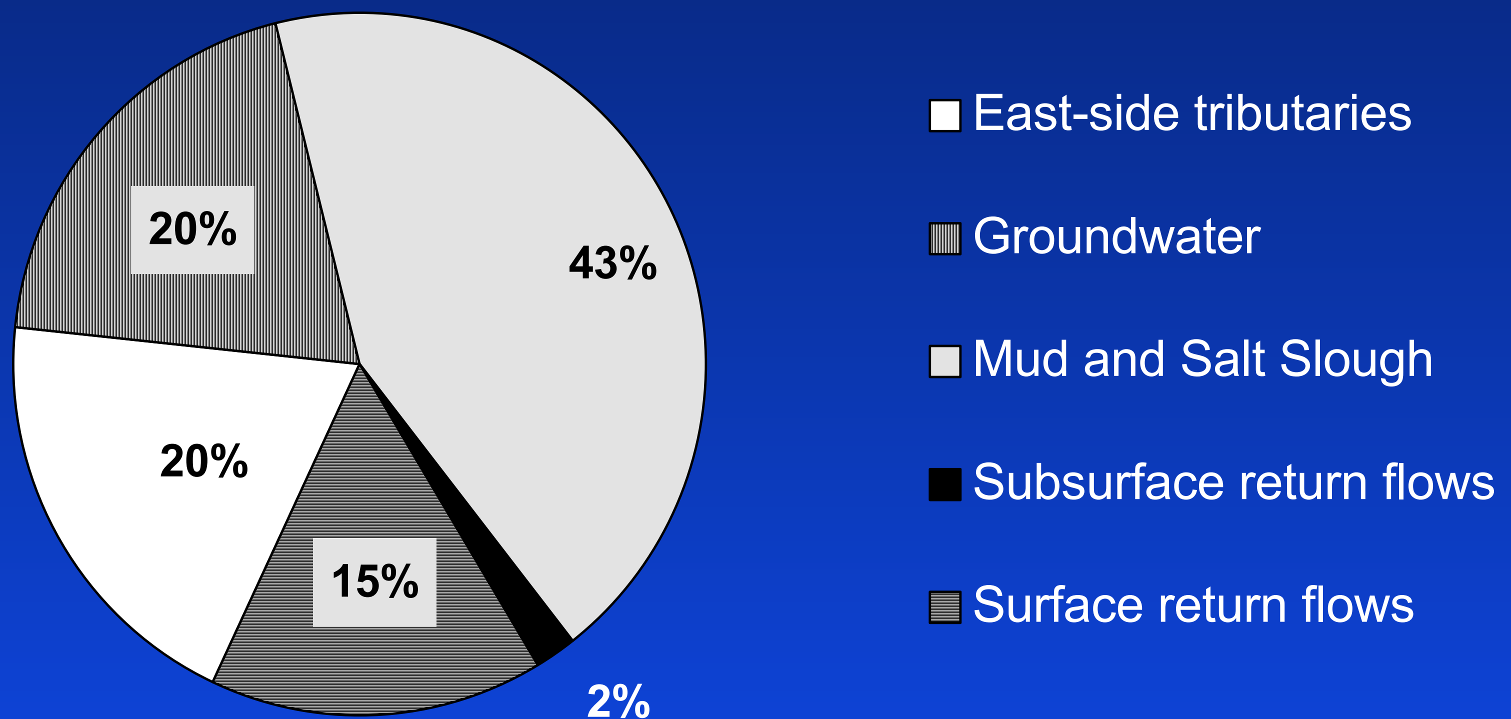


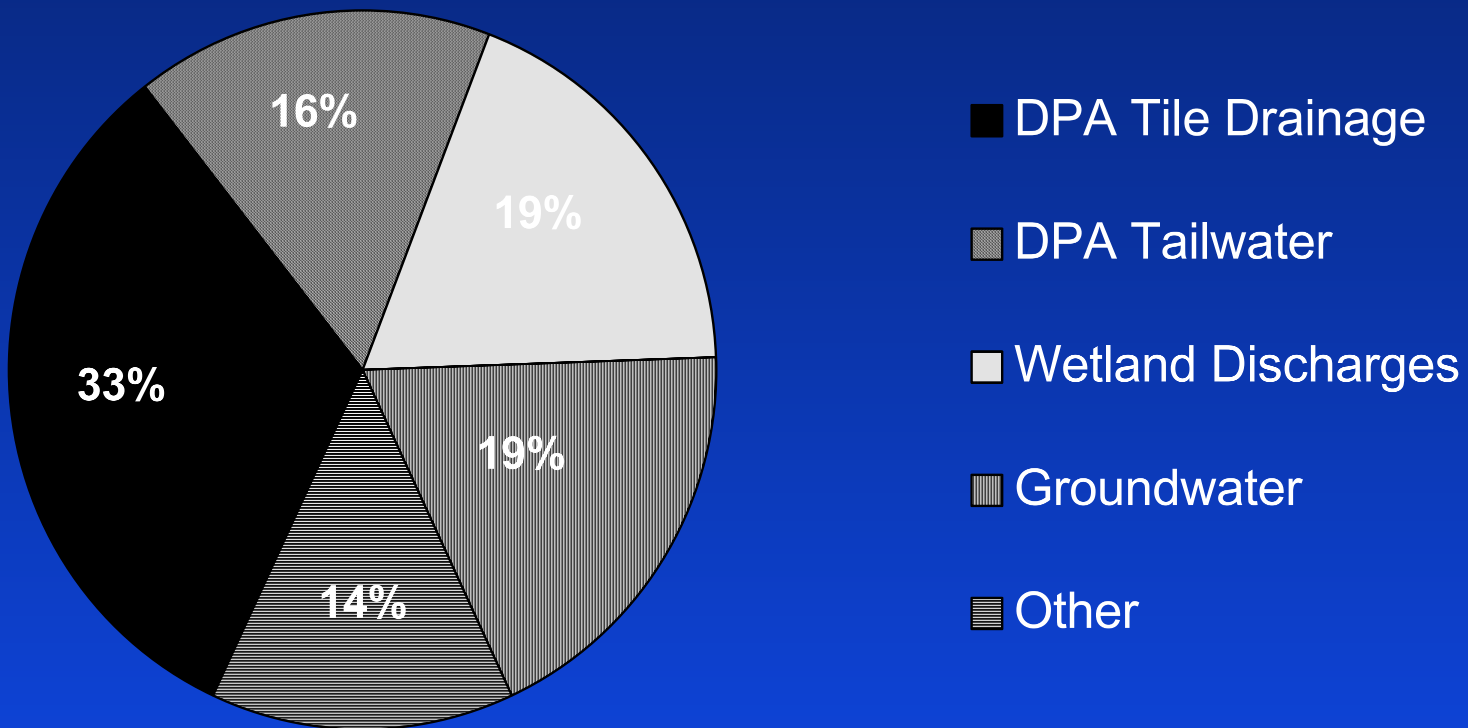
Figure II-5 Lower San Joaquin River TDS Load



Mean Annual Loading of TDS to SJR for WY 85 to 95: 1 million tons

Basis: Historical and SJRIO model data

Figure II-6 Mud and Salt Slough TDS Load



Mean Annual Loading of TDS to Sloughs for WY 85 to 95: 430,000 tons
Basis: Spreadsheet analysis of DSA discharges and wetland operations

Figure II-7 Lower San Joaquin River TDS Load

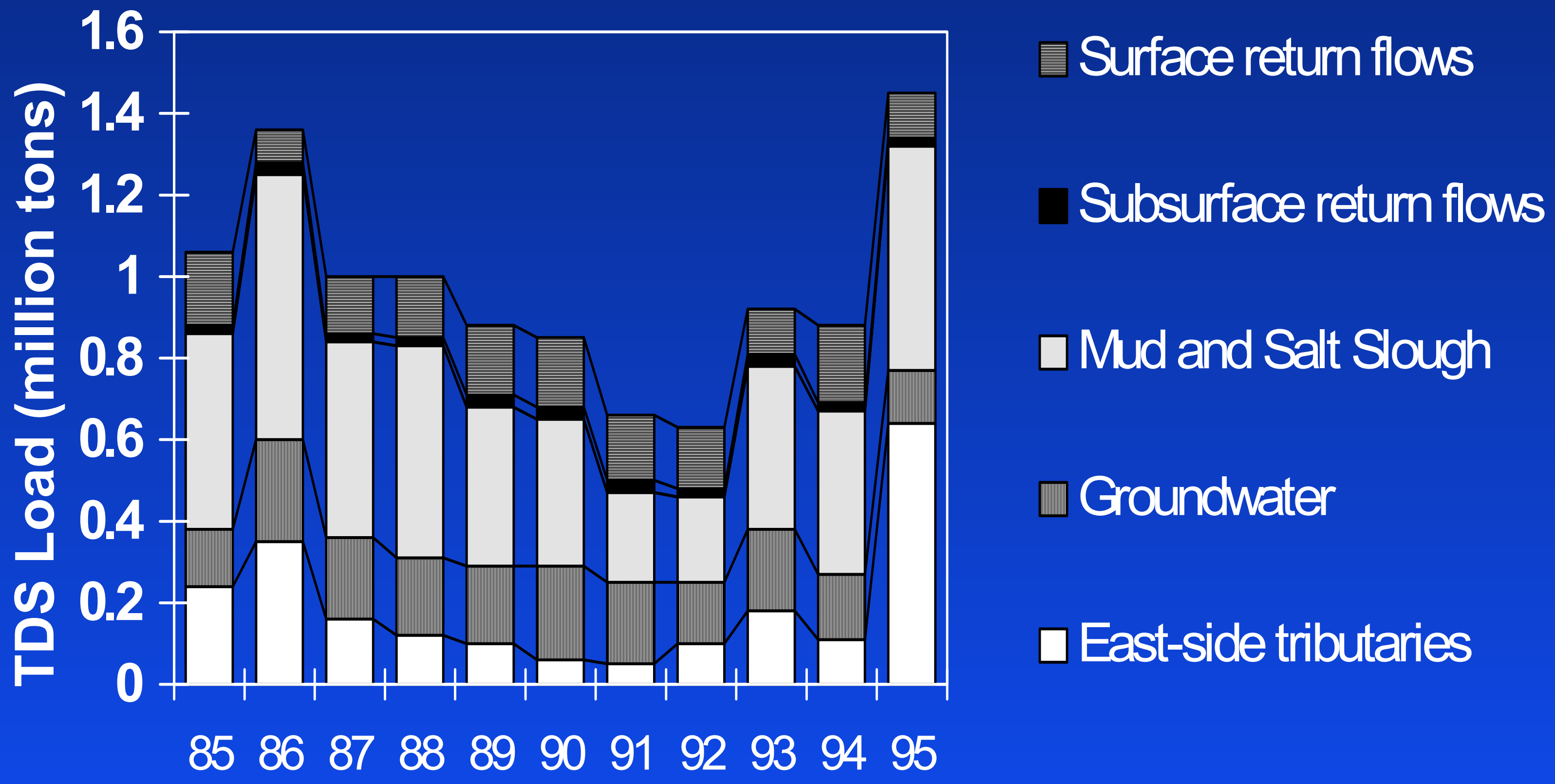
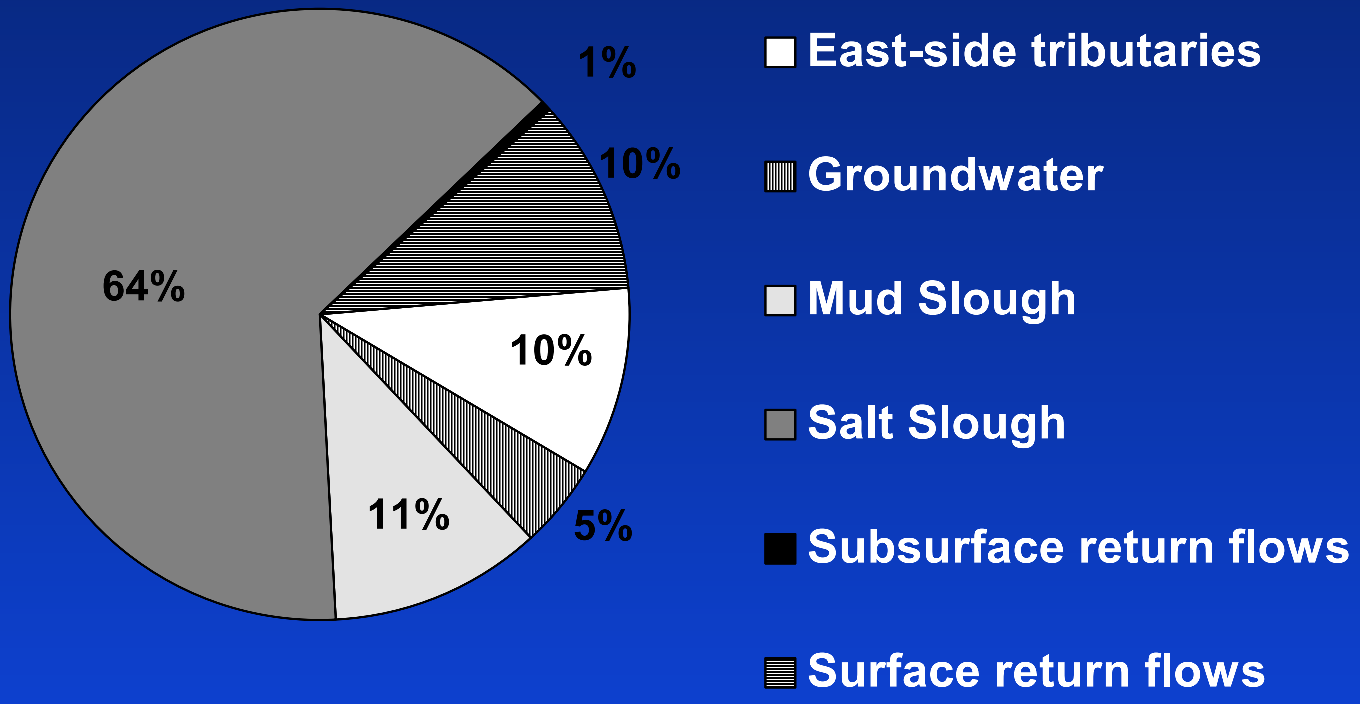


Figure II-8 Lower San Joaquin River Selenium Load



Mean Annual Loading of Selenium to SJR for WY 85 to 95: 10,000 pounds

Based on combination of historical and SJRIO model data

Figure II-9 Lower San Joaquin River Selenium Load

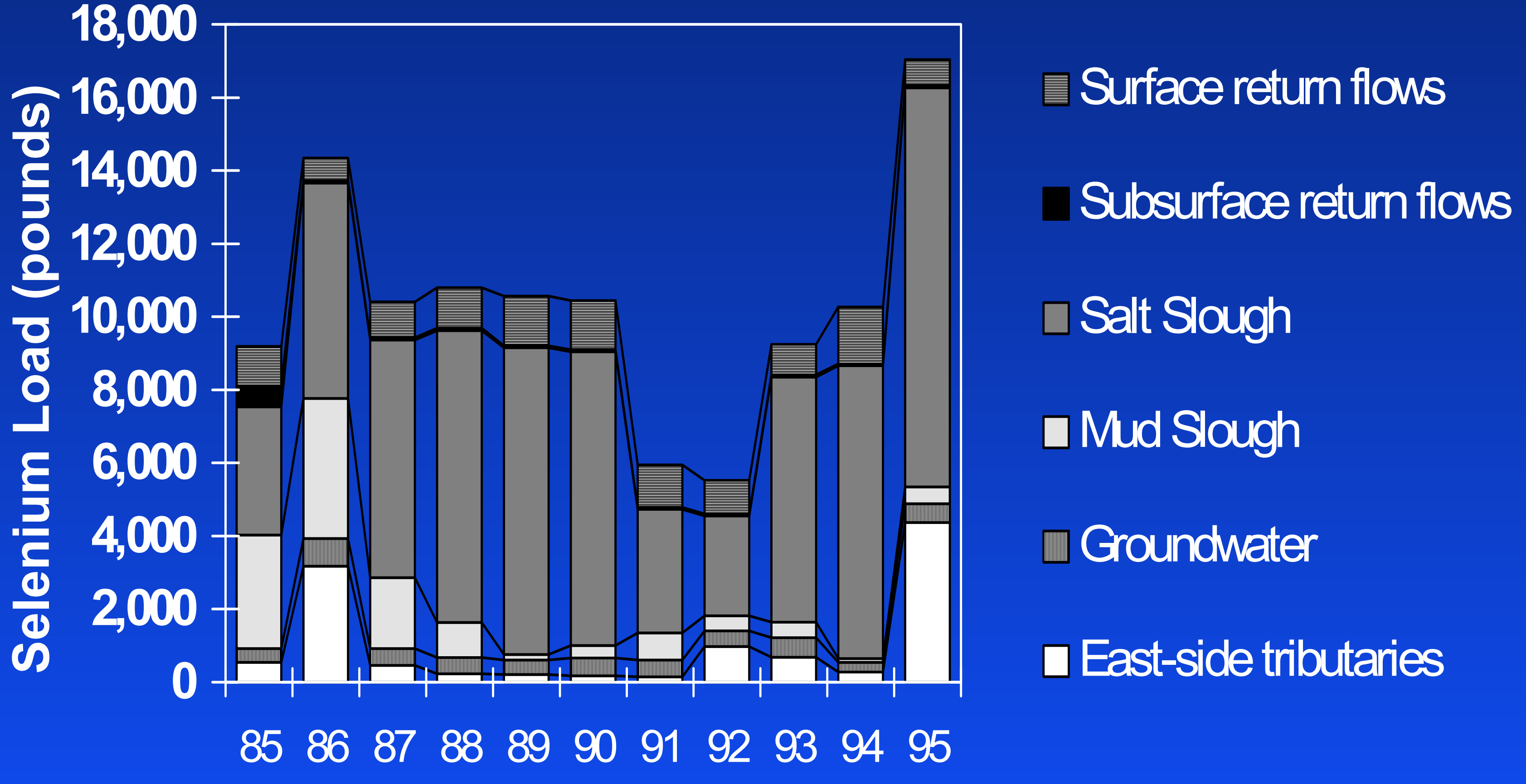


Figure II-10 Lower San Joaquin River Mean Annual Discharge, TDS, and Selenium Loads

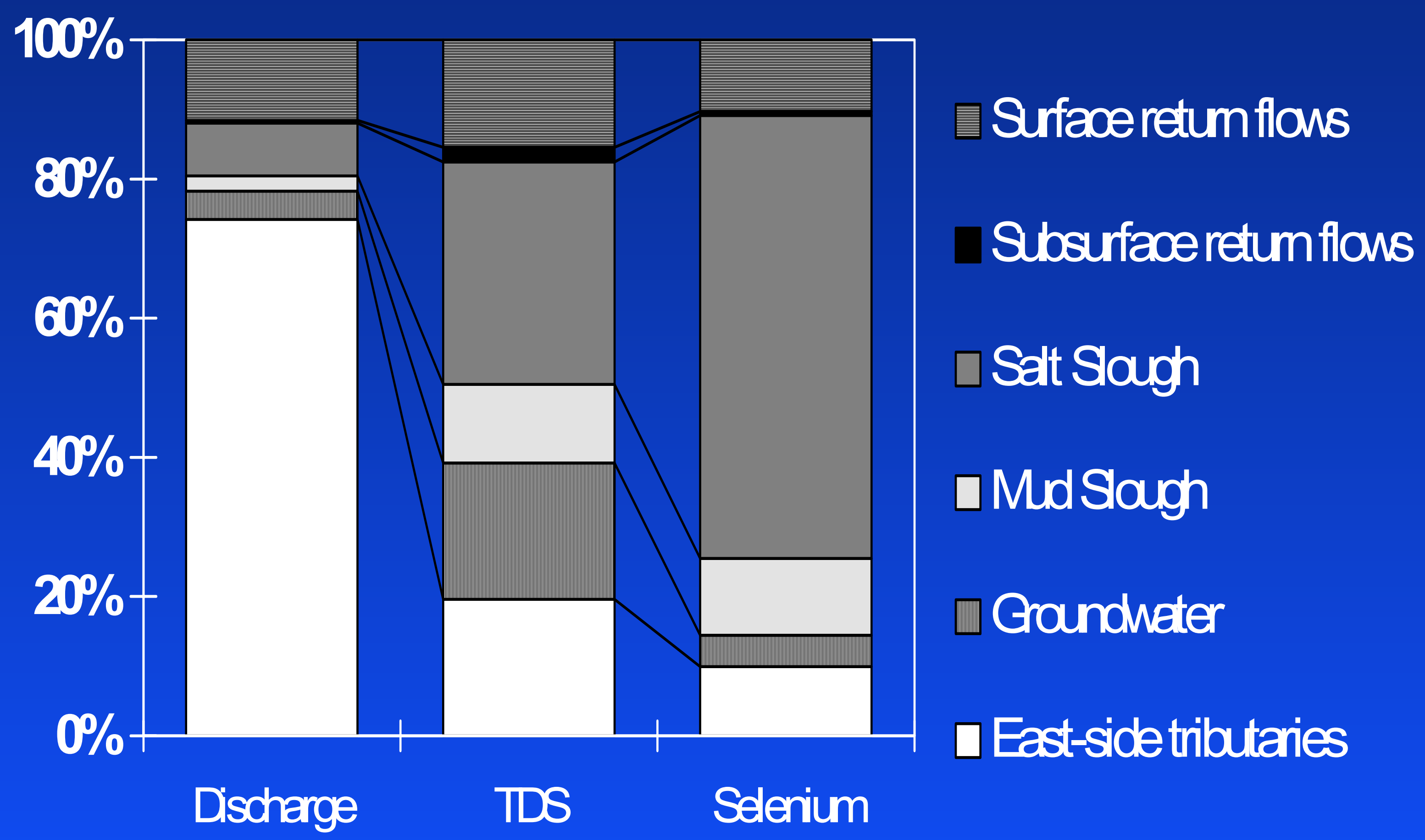


Figure II-11 Annual Discharge from the Drainage Project Area, Grassland Watershed, and the San Joaquin River (SJR) at Crows Landing and Vernalis, Water Years 1986 through 1997 (Chilcott et al. 1998a)

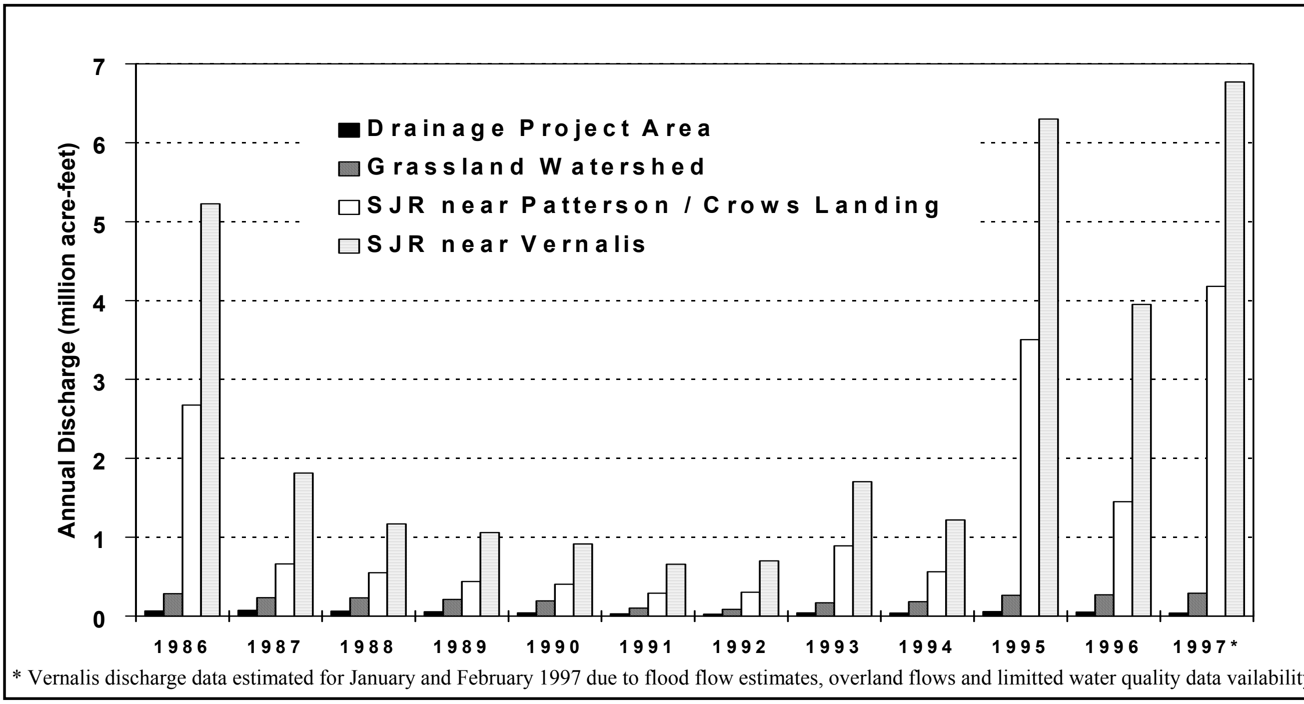
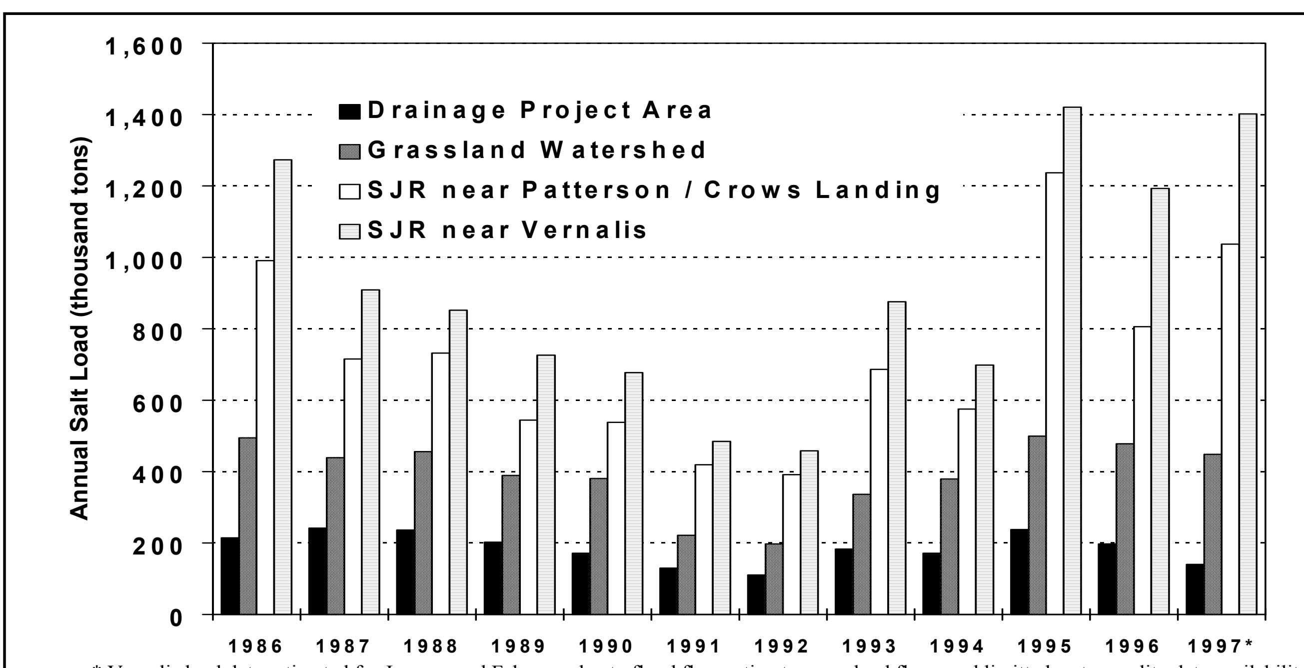


Figure II-12 Annual Salt Load from the Drainage Project Area, Grassland Watershed, and the San Joaquin River at Crows Landing and Vernalis, Water Years 1986 through 1997 (Chilcott et al. 1998a)



filename: figsjr2.ppt

Figure II-13 Annual Boron Load from the Drainage Project Area, Grassland Watershed, and the San Joaquin River (SJR) at Crows Landing and Vernalis, Water Years 1986 through 1997 (Chilcott et al. 1998a)

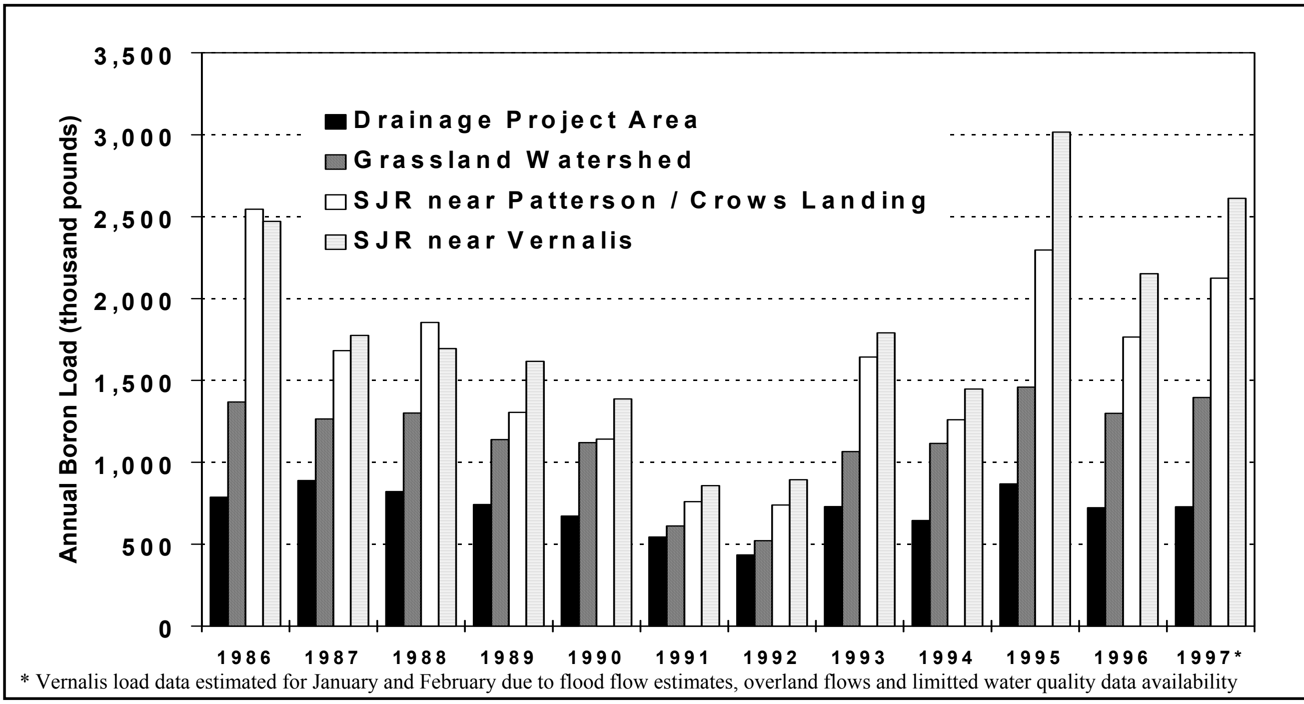
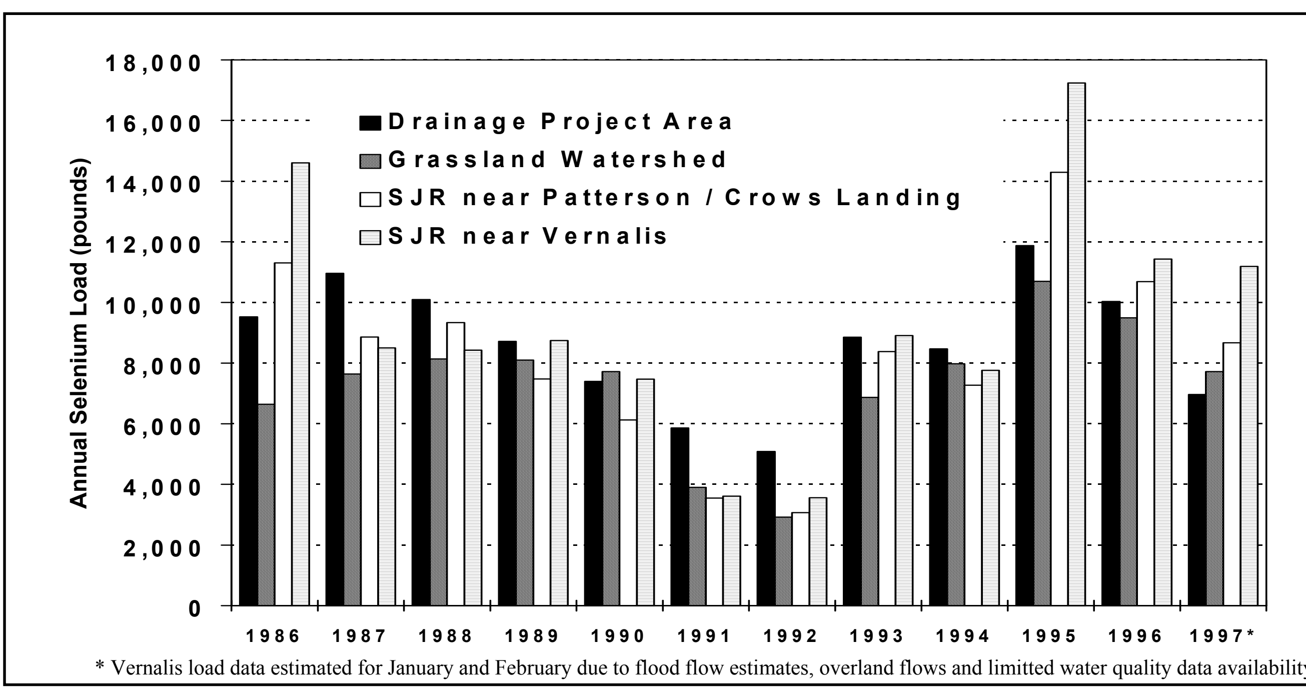


Figure II-14. Annual Selenium Load from the Drainage Project Area, Grassland Watershed, and the San Joaquin River at Crows Landing and Vernalis, Water Years 1986 through 1997 (Chilcott et al. 1998a)



filename: figsjr.ppt

Table IV-1 Lower San Joaquin River Beneficial Uses from Sack Dam to the Delta as Designated in the Water Quality Control Plan

(Source: *Water Quality Control Plan, Central Valley, San Joaquin River Basin* ; CVRWQCB 1994; 1996)

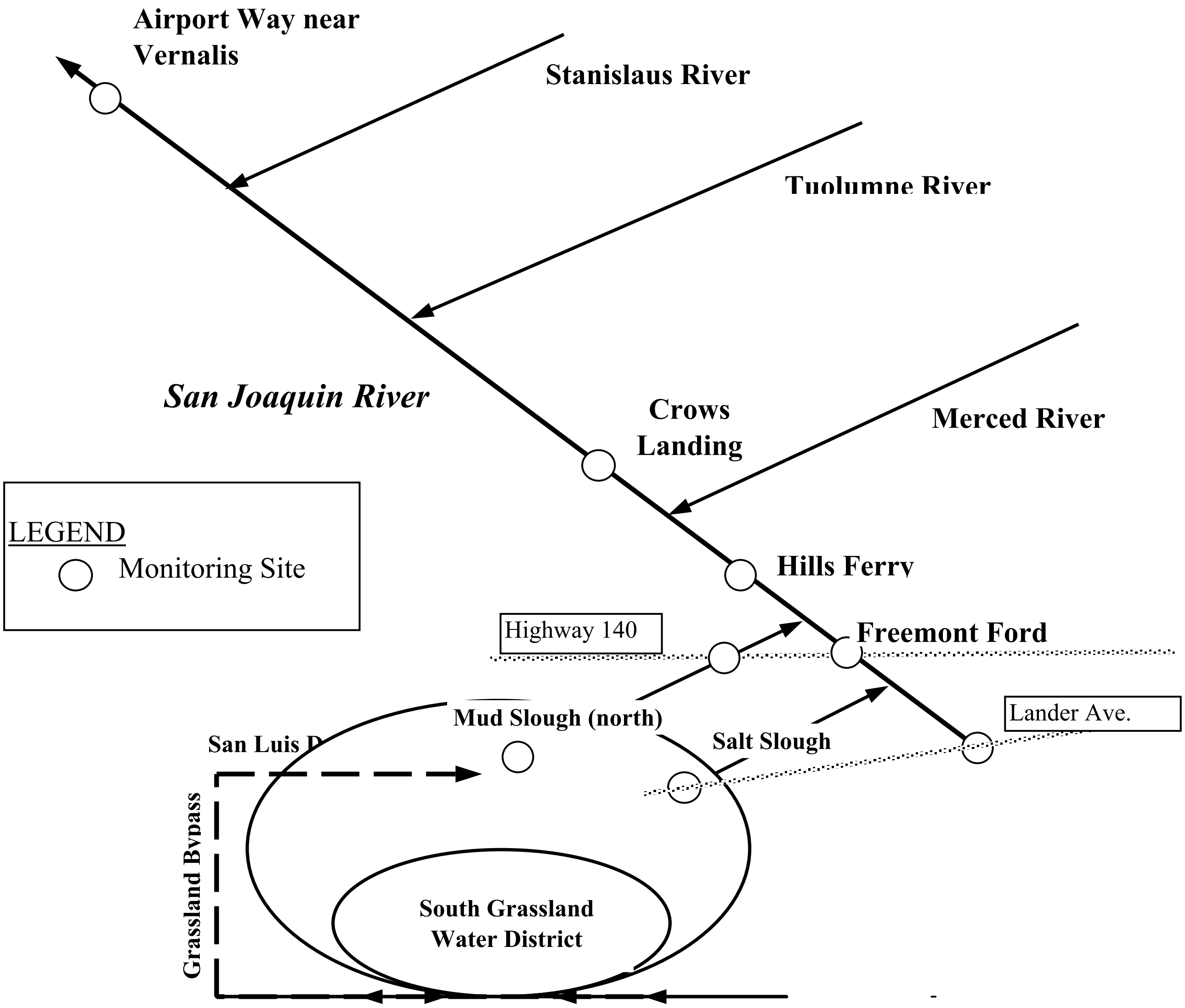
SURFACE WATER BODIES (1)			AGRI-CULTURE		INDUSTRY			RECREATION			FRESHWATER HABITAT (2)		MIGRATION		SPAWNING			
		MUN	AGR		PROC	IND	POW	REC-1		REC-2	WARM	COLD	MIGR		SPWN		WILD	NAV
		MUNICIPAL AND DOMESTIC SUPPLY	IRRIGATION	STOCK WATERING	PROCESS	SERVICE SUPPLY	POWER	CONTACT	CANOEING AND RAFTING (1)	OTHER NONCONTACT	WARM	COLD	WARM (3)	COLD (4)	WARM (3)	COLD (4)	WILDLIFE HABITAT	NAVIGATION
MENDOTA DAM TO SACK DAM		P	E	E	E			E	E	E	E		E	E	E	P	E	
SACK DAM TO MOUTH OF MERCED RIVER		P	E	E	E			E	E	E	E		E	E	E	P	E	
MOUTH OF MERCED RIVER TO VERNALIS		P	E	E	E			E	E	E	E		E	E	E		E	
SACRAMENTO/SAN JOAQUIN DELTA		E	E	E	E	E		E		E	E	E	E	E	E		E	E
SALT SLOUGH			E	E				E		E	E				E		E	
MUD SLOUGH (NORTH)			L(5)	E				E		E	E				E		E	

LEGEND
E=EXISTING BENEFICIAL USES
P=POTENTIAL BENEFICIAL USES
L=LIMITED

(1) Shown for streams and rivers only with the implication that certain flows are required for this beneficial use.
(2) Resident does not include anadromous. Any Segments with both COLD and WARM beneficial use designations will be considered COLD water bodies for the application of water quality objectives.
(3) Striped bass, sturgeon, and shad.
(4) Salmon and steelhead.
(5) Elevated natural salt and boron concentrations may limit this use to irrigation of salt and boron tolerant crops. Also low flow conditions may limit use.

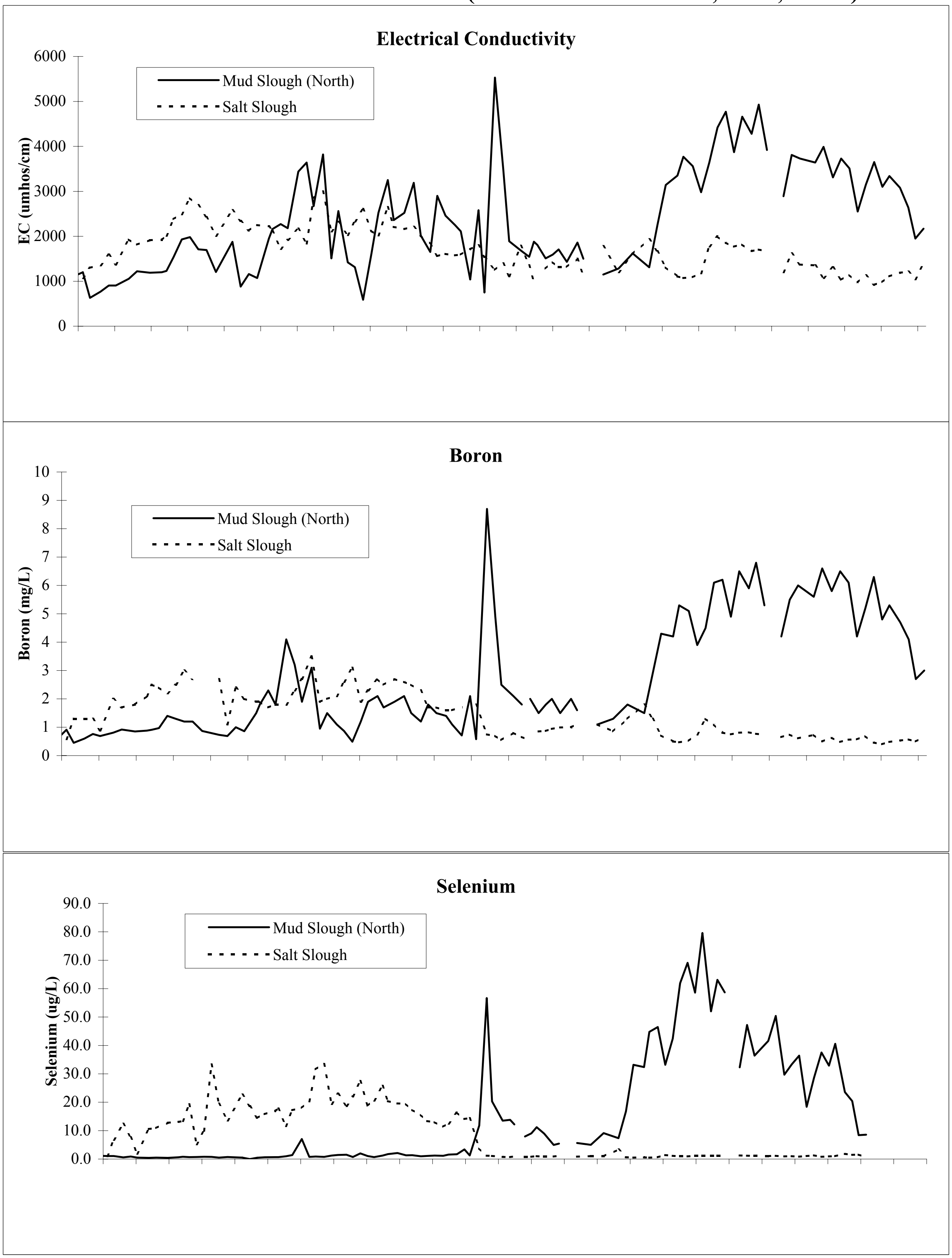
filename: sjvdip\benutab.xls 5/20/98

Figure VI-1 Schematic of Selected Monitoring Sites in the Lower San Joaquin River Basin



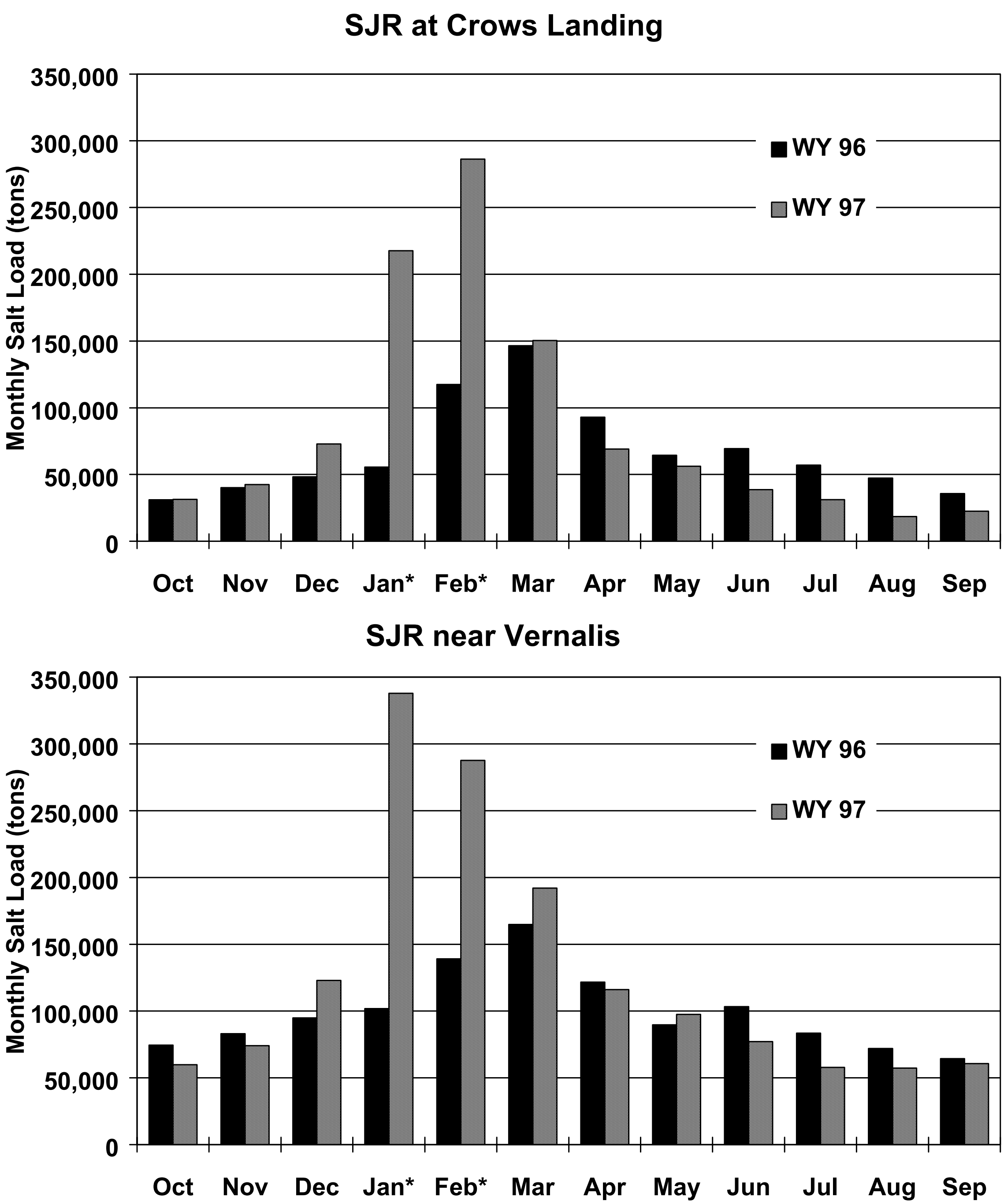
filename: monmap2.xls

Figure VI-2. Comparison of Electrical Conductivity, Boron and Selenium at Salt Slough and Mud Slough (North) Downstream of the San Luis Drain: Water Years 1996 and 1997 (modified from Chilcott, *et al.*, 1998b)



filename: mudsalt.xls

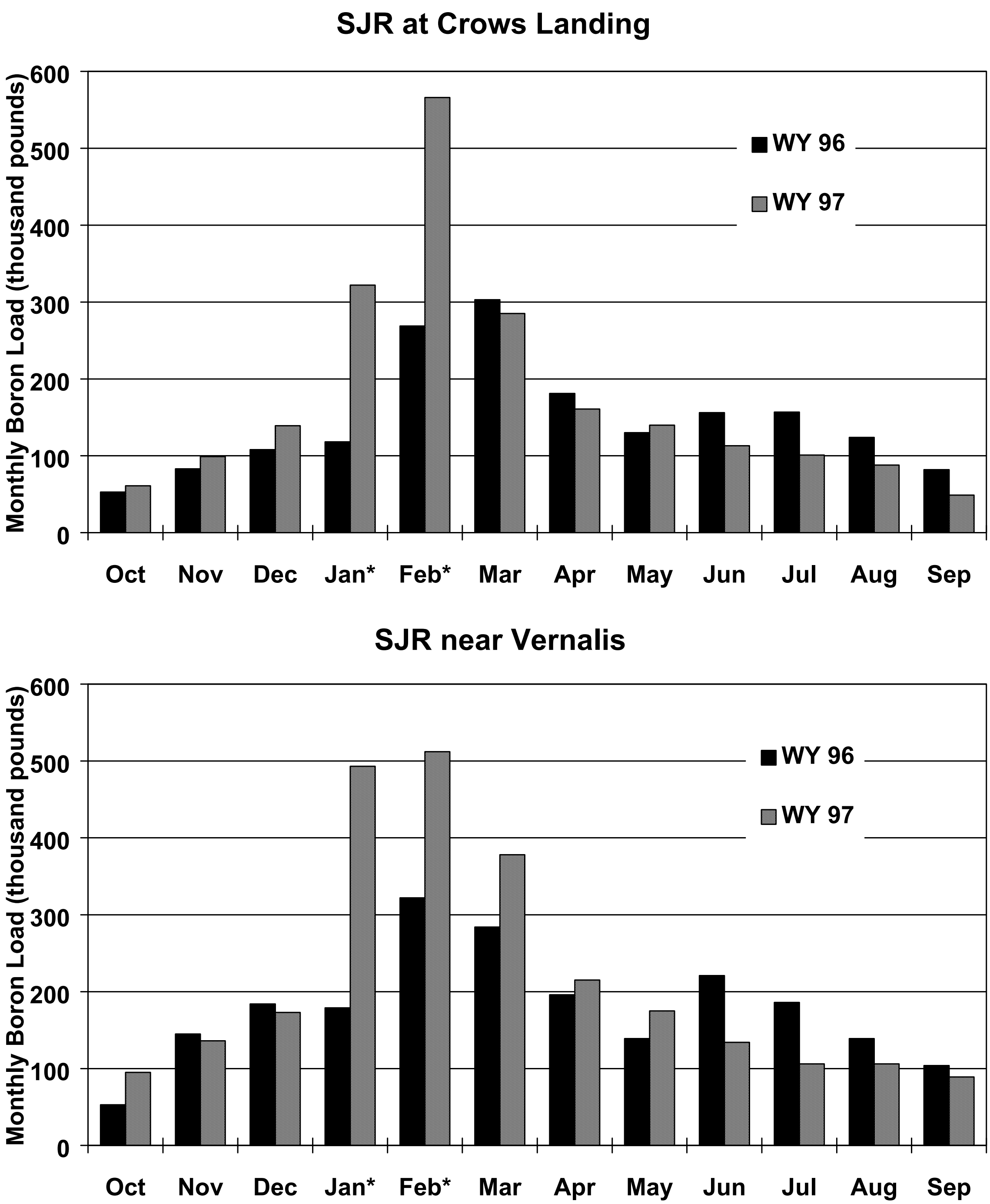
Figure IV-4 Monthly Salt Loads Measured in the San Joaquin River (SJR) at Crows Landing and Vernalis, Water Years (WYs) 1996 and 1997 (Chilcott et al. 1998a)



* load data estimated for January and February 1997 due to flood flow estimates, overland flows and limitted water quality data availability

filename: figsjr2.ppt

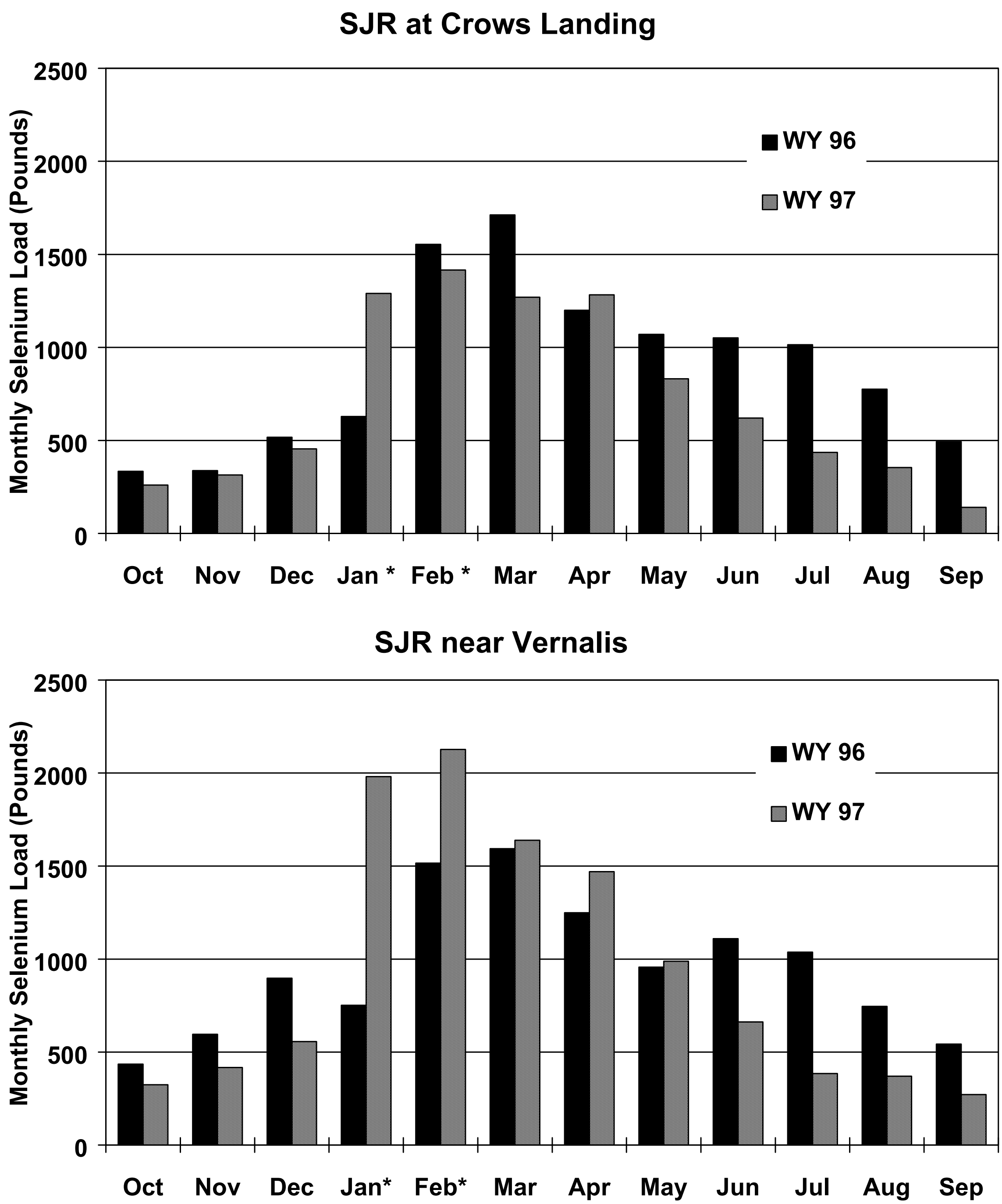
Figure VI-5 Monthly Boron Loads Measured in the San Joaquin River (SJR) at Crows Landing and Vernalis, Water Years (WYs) 1996 and 1997
(Chilcott et al. 1998a)



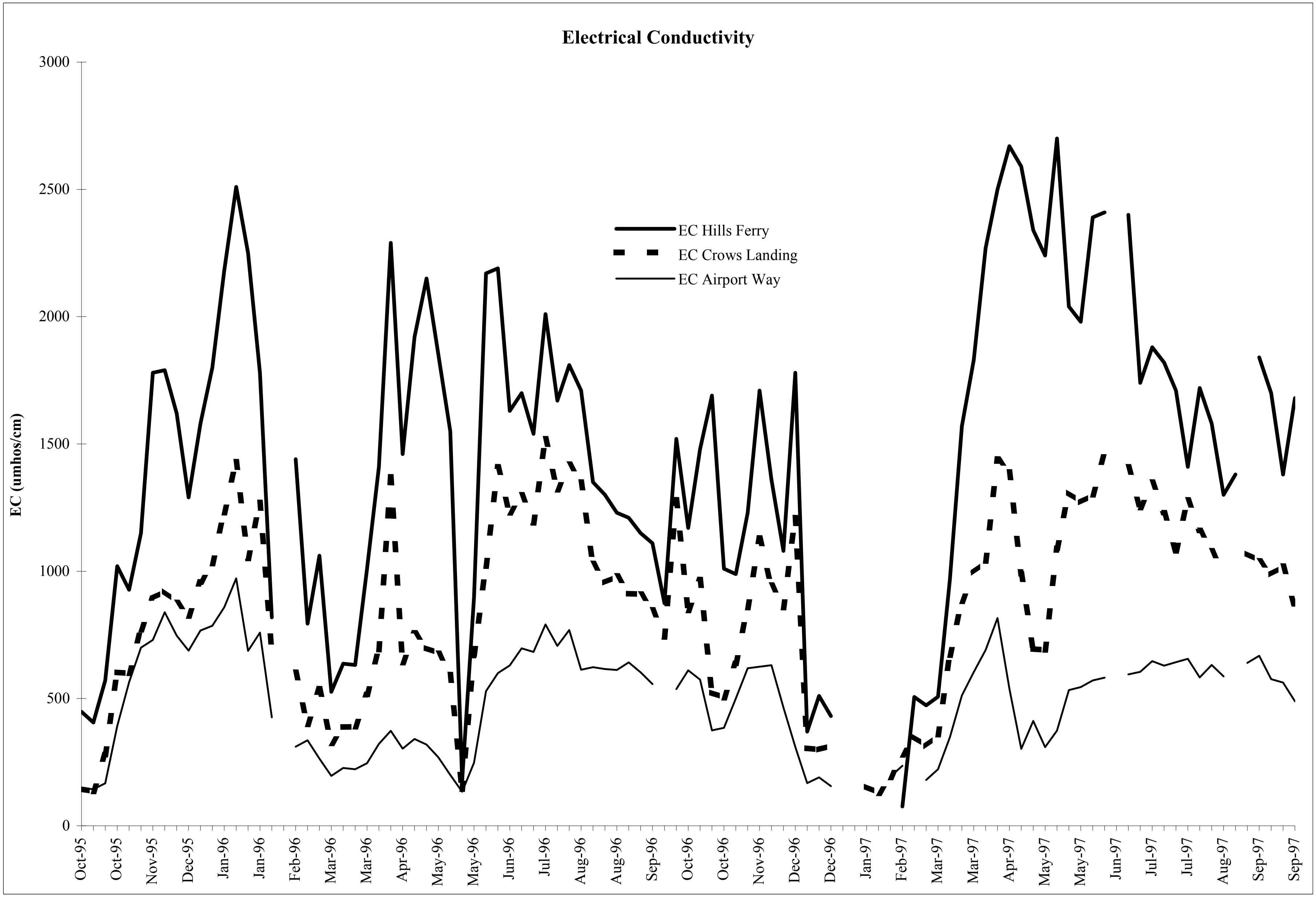
* load data estimated for January and February 1997 due to flood flow estimates, overland flows and limitted water quality data availability

filename: figsjr2.ppt

Figure VI-6 Monthly Selenium Loads Measured in the San Joaquin River (SJR) at Crows Landing and Vernalis, Water Years (WYs) 1996 and 1997 (Chilcott et al. 1998a)



* load data estimated for January and February 1997 due to flood flow estimates, overland flows and limitted water quality data availability filename: figsjr2.ppt


Electrical Conductivity
EC (umhos/cm)
3000
2500
2000
1500
1000
500
0
EC Hills Ferry
EC Crows Landing
EC Airport Way
Oct-95
Oct-95
Nov-95
Dec-95
Jan-96
Jan-96
Feb-96
Mar-96
Mar-96
Apr-96
May-96
May-96
Jun-96
Jul-96
Aug-96
Aug-96
Sep-96
Oct-96
Oct-96
Nov-96
Dec-96
Dec-96
Jan-97
Feb-97
Mar-97
Mar-97
Apr-97
May-97
May-97
Jun-97
Jul-97
Jul-97
Aug-97
Sep-97
Sep-97